Land Use and Carbon Dynamics in Woody Ecosystems

by

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Publications included in this thesis

This thesis contains eight manuscripts that have been published in peer reviewed literature. They have been incorporated as Chapters 2–4, 7–10, and Appendices IV & V, and were modified under supervision during the candidature.

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Abstract

Humanity has not yet been able to contain its influence on global average temperatures nor on the carry-over effects of that to the biosphere, namely climate change initiated by our carbon emissions, and positive feedback on temperature is likely to increase. The contribution played by our land use change (LUC) is millennia-old and annually it is currently comparable to that of fossil fuel usage, but it is less precisely known. The key aim of this thesis is to improve the knowledge about carbon emissions from LUC to the atmosphere so that we can control them better ('climate change mitigation'), and in order to facilitate better climate change modelling.

Extremes of intensity and expanse of change in carbon, accompanying LUC, were studied using carbon-dense forests and rangelands respectively. The dynamics were reviewed and re-assessed, and previously discounted carbon pools were quantified. Tools used in the analysis included conceptualisation of carbon dynamics, standard forest mensuration, remote sensing and GIS, soil data collection and carbon assay, terrestrial photogrammetry using innovative software, formulae development, and mathematical modelling including forecasting of carbon flux. Most data were from Australia but the concepts applied, theories developed and thematic results are applicable globally. For intense LUC the focus was on the more carbon-dense forests, especially those in Tasmania which were the major feedstock of the Pacific rim's hardwood pulpwood market until 2012. For expansive areas the Australian commercially grazed rangeland was studied and two case studies were the States of Queensland and New South Wales. In those, LUC had recently been more intensive, with accompanying greater public scrutiny and consequently more data availability.

The findings will contribute towards more precise estimates of carbon fluxes for emission trading schemes, national reports and for climate change modelling from several millennia in the past to two in the future. There were several main areas of knowledge enhancement:

1) increased quantification of anthropogenic influence on C flux in woody ecosystems accompanying logging and rangeland grazing, past and future.

2) revealing that the half-lives of wood products need to be increased by nearly an order of magnitude to not lose nearly half the carbon of primary forests when they are converted to secondary forests on harvesting cycles;
3) increased formulaic relations between natural phenomena for use in modelling carbon dynamics;
4) showed how a comprehensive time-space (4D) context of carbon dynamics in woody ecosystems can reveal both higher carbon emissions and higher carbon sequestration for LUC or ecosystem recovery respectively, and how it provides an overarching explanation to relieve points of contention by (e.g. for woody-thickening, deforestation; and forest logging);
5) increased understanding of SOC distribution and change (e.g. development of a robust method to calculate cumulative SOC down a diverse soil profile; finding associated with large trees ~6% more SOC than earlier tallied, the same trees targeted for logging;
6) improved experimental methods, e.g.: use of Photoscan software to quantify large trees, and remote sensing/GIS to locate carbon stock benchmarks in degraded rangeland;
7) discovery of a new natural substance (cunnite) constituting a novel but small carbon pool; and
8) informative, diagrammatic portrayal of a sequence of relationships between land-use, woody ecosystems, and climate, in the carbon cycle.

To help increase knowledge it is imperative, for carbon accounting of both forest logging and rangeland commercial grazing, that greater data transfer occurs between industry, government departments and the scientific community.

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Table of Contents

Chapter 1 Introduction

1.1 Relation of vegetation disturbance by humans to anthropogenic climate change

1.1.1 Historical effects

Since establishment of the IPCC (Intergovernmental Panel on Climate Change) in 1988 there has been enthusiasm in some quarters to address anthropogenic climate change by attenuating growth in atmospheric greenhouse gasses (GHG)— 'climate change mitigation'. Nevertheless, annual anthropogenic carbon (C) efflux continues to increase and urgency has arisen owing to the likelihood that temperature rise will not be kept below 2° C (Gasser et al., 2015), and the increasingly obvious positive feedback from climate change (e.g. snow (albedo) loss, more-frequent severe forest fires, more severe dry-period impacts on biomass).

Fertilisation by CO_2 and longer growing season [in cooler climates] were predicted to increase biomass (and may have done so previously in Australia), however, in the last decade there has been increasing evidence of stress effects on forests [from climate change] either limiting growth or reducing standing biomass in many non-polar regions (e.g. Barber et al., 2000; Calder and Kirkpatrick, 2008; Allen et al., 2009; Wallace et al., 2009; Carnicer et al., 2011; Hofmockel et al., 2011; Piao et al., 2011a) through increased severity of droughts and fire, which in turn reduces net biomass (i.e. positive feedback). Also, positive feedback from climate change will increasingly detract from the carbon stores of some forests as climate change progresses (Houghton, 1997; Kashian et al., 2006; Warszawski et al., 2013).

Anthropogenic land use can be called by different names in the field of carbon accounting, depending on the degree to which the main vegetation type is altered.

Example acronyms are: LULUCF (land use, land-use change and forestry, IPCC (2000)), LUCC (land use and land cover change, Gamboa and Galicia (2011)), LCC (land cover change) and LMC (land management change). LMC means no major change in the type of item constituting the land cover (i.e. zero LCC) but signifies change in its concentration, e.g. through logging intensification (Luyssaert et al., 2014). In this thesis I use the acronym LUC, meaning land use change including, dramatic change in use such as from logged-forest to wheat farming, or more subtle changes like introduction of commercial livestock without tree felling, change from the non-anthropogenic fire regime (e.g. to prescribed burning or arson), or change in logging intensity. The reason for this choice is that they can all alter the carbon dynamics and thereby affect climate (IPCC, 2000; Luyssaert et al., 2014).

Although annual emissions from fossil fuel usage plus cement manufacture are higher than those from anthropogenic land-use change (LUC), cumulative emissions over millennia from LUC (Edney et al., 1990; Caseldine and Hatton, 1993; Rudiman, 2003; Salinger, 2007; Olofsson and Hickler, 2008; Metz, 2009; Pinter et al., 2011; He et al., 2014) outweigh those from fossil fuels, with data in Kaplan et al. (2010) and Le Quéré et al. (2015) showing a ratio of 1.25:1. Examples of LUC with net anthropogenic reduction in woody biomass from the late Pleistocene to the present day are: forest removal or degradation for pasture, urbanisation, and crops (agricultural, energy, and forestry, including reduction from primary[1] to secondary forests or plantations). Around 40–60% of anthropogenic emissions since pre-industrial times remain in the atmosphere (House et al., 2002; Elmegreen Rafelski et al., 2009) and continue to contribute to climate change. Future anthropogenic impact from land-use may emit a further 100 Pg of carbon (C) and constitute 9–22% of the total anthropogenic increase in CO_2 to 2100 (Schimel, 1995; Sitch et al., 2005). The term ‘forest degradation’ is used here to refer to both gradual attrition of a forest’s

[1] Regarding use of the terms ‘primary forest’ or ‘oldgrowth’ forest I follow Wirth et al. (2009) with primary forest being ‘…stands that have been subject to very low levels of human impact for an extended period of time. This includes stands of any age and time since disturbance.’ Whereas for oldgrowth forest: ‘…stands that have reached a certain age or late successional stage and that may or may not have been impacted by humans. For example, old-growth stands may originate from a planted stand developing after clear-cut.’ Much oldgrowth in Australian TOF is thus primary forest.

woody biomass (whether through erosion from overgrazing, thinning, or reduced water infiltration from weeds or loss of debris etc.) or immediate deforestation, such as in ‘land clearing’ and ‘fence-line clearing’.

Emissions from fossil fuel usage overtook those from land-use in 1959(±7) (Woodwell et al., 1983). However, from more recent data (Houghton, 2008; Steffen, 2009) and by assigning polynomials to emission trends (Figure 1-1) it appears that it was in 1913(±10) that the C efflux from fossil fuel combustion, exceeded that from LUC. Additionally, when decomposition of wood-products and change in soil organic carbon (ΔSOC) from land-use are included then fossil fuel emissions only overtook emissions from land-use in 1988(±20). Fossil fuel emissions may have only surpassed those from land-use even more recently than that, as Houghton (2008) had assumed zero ΔSOC with timber harvesting, which, from more extensive experimentation, seems less likely (e.g. Diochon et al., 2009; Zummo and Friedland, 2011; Clarke et al., 2015, Chapter 7).

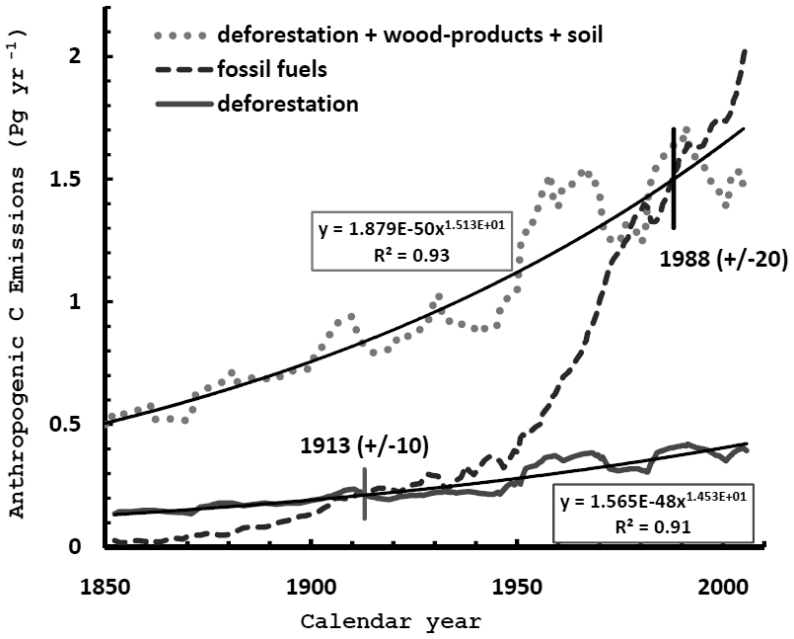


Figure 1-1 Comparison of major sources of anthropogenic C emissions.
Data sources: deforestation and fossil fuels, (Steffen, 2009); deforestation plus wood-products and soil change, (Houghton, 2008). (N.B. Houghton assumed zero change in soil carbon with timber harvesting.)

As 25–50% of anthropogenic emissions remain in the atmosphere for 500–10,000 years (Archer et al., 2009; Eby et al., 2009; Houghton et al., 1994) then improved accounting for both historic and future emissions will aid climate-change modelling. Uncertainty in anthropogenic emissions contributes to ~1.5º C uncertainty in the temperature effect of climate change to 2100 (Meir et al. 2006). For example, the effect from uncounted LUC emissions masks the measurement of reduced C sink efficiency (Gloor et al., 2010). Uncertainty in LUC emissions remain the highest in the global carbon budget and this is mostly due to uncertainty in change in soil organic carbon (SOC) and roots (Kim and Kirschbaum, 2014; Luo et al., 2015). Emissions from both biomass and SOC, due to LUC, contribute to climate change, and in places there is feedback to further land degradation of drylands (Sivakumar, 2007). Thus, for the purposes of C management, LUC influences on SOC are significant and worthy of scrutiny, especially as their emissions can be less immediate (and hence less easily measured) than those from fossil fuel combustion.

1.1.2 Calculable effects

The millennia-long impact on the atmosphere from our LUC emissions, and the potential for re-sequestration, can be calculated from the carbon stocks of soil and trees in past and present primary forests (Dixon et al., 1994; Rhemtulla et al., 2009; Houghton et al., 2012; Luo et al., 2015). Soil organic carbon (SOC) is a substantial terrestrial C pool, often positively correlated with plant biomass (Su and Zhao, 2003; Harms et al., 2005; Wheeler et al., 2007; Roxburgh, 2009; Frank et al., 2012), primarily resulting from root turnover and litterfall. Thus, in the long-term, decline in biomass would be expected to causes decline in SOC— a subsidiary LUC emission. Current knowledge gaps and low precision for industrial use of natural carbon pools such as wood and soil prevent better climate-change modelling and through societal mechanisms, they add to the momentum of avoiding climate change mitigation.

The amount of C per unit area (Mg ha^{-1}) is the carbon two-dimensional (vertically projected) ‘density’, or unit-area carbon stock. Change in carbon density is via a

flux— mass movement per unit area per unit time. Of the three major organic C pools: vegetation biomass (above and below ground), SOC, and debris (e.g. coarse woody debris), vegetation is the easiest to measure in the field or by remote-sensing, and the fastest to be influenced by LUC.

Temperate forests currently occupy ~600 Mha worldwide, with most directly impacted by humans (Heath et al. 1993). Forestry is one of the major contemporary industrial uses of forests, and accounting of forestry's industrial carbon emissions is important not only to determine the carbon fluxes accompanying past and future industrial activity but also in climate change modelling. The area of tall open-forest (TOF) in Australia has principally decreased since Europeans first settled here (in ~1750) as the TOFs were either 'mined' for timber or simply cleared for agriculture (Kirkpatrick, 1986; Kirkpatrick, 1994) plus clearing for mineral mining and urbanisation (reducing temperate forest from 113 Mha to 55 Mha). A net, gradual SOC emission is generally observed or forecast for the conversion of primary-forest to harvested secondary-forest (Christophel et al., 2013), but minimal change is often reported over shorter durations (e.g. Hopmans et al., 2005; Zummo and Friedland, 2011, Chapter 7). Accounting of forestry's industrial carbon emissions is important not only to determine the carbon fluxes accompanying present and future industrial activity but also in climate change modelling. That industry's carbon balance is examined in this study.

Although a net annual C sink is reported for the Australian forestry industries (Department of Environment, 2015) its calculation ignores some emissions from the original conversion of primary TOF to secondary forest and plantation, prior to instigation of emissions reporting; for example: the original intense silvicultural burn (30–50% of aboveground biomass, Ximenes et al. (2008a)), decomposing roots of the primary forest, and the long-term decrease in soil carbon accompanying the lower, temporal-average aboveground biomass (AGB) (e.g. Roxburgh, 2009). Additionally, emissions from wood-products in landfill are counted in the 'waste sector' rather than for the forest industry.

When considering soil organic carbon for mesic forest or rangeland, in order to define the time span for long-term change, one can defer to the time taken for

perturbations in mineral soil organic carbon stocks to approach an asymptote after land-use or land-cover change. Long-term is therefore at least 1,000 years and more towards 2,000 years, i.e. the time taken for SOC to equilibrate under the new biomass regime (e.g. Liski et al., 1998; Hibbard et al., 2003). Short-term is the time taken for the biomass of an even-aged forest stand to approach an asymptote, e.g. ~70 years, an order of magnitude less than long-term.

Spatial heterogeneity of SOC when measuring change in SOC (ΔSOC) with LUC has also been insufficiently represented. When considering how best to observe ΔSOC associated with logging of primary forest, that has a post-felling silvicultural burn, the ideal experimental setup would be when the fire escapes into neighbouring unlogged forest (an 'escaped regeneration burn'). The pairs of sites, either side of the logging boundary can then be reasonably assumed to have had equivalent environment, previous site history and forest age, and therefore similar legacy carbon (Harmon, 2001), except for the recent logging. A remaining difficulty is then to ensure spatially equivalent sampling before and after logging, and fire. Appropriate sampling is unlikely if, prior to logging, the areas near and under large trees and large logs are avoided due to the physical difficulty of sampling, which is common practice.

Sampling under and adjacent to tree trunks appears to be extremely rare in temperate, boreal or tropical forests. For example it was not done in Bockheim (1977); Entry and Emmingham (1998); Fang et al. (2010); Dietrich (2012); Fedrigo et al. (2014) or Schrumpf et al. (2014). Tree trunks, roots and soil are highly disturbed during some silvicultural techniques such as clearfell-and-burn, making the landscape much more physically accessible, and there is redistribution of soil (Ellis and Graley, 1983; Rab, 1996; Pennington et al., 2001). Mature trees can leave a long-term spatial imprint in the soil (Døckersmith et al., 1999; Phillips and Marion, 2005) and SOC can be more concentrated closer to trees, either through thicker humus layers or higher C concentration in the mineral soil, or both (Lutz, 1960; Liski, 1995; Throop and Archer, 2008; Rossetti et al., 2015).

However, SOC sampling under large tree trunks has been avoided due to physical constraints (e.g. in Bockheim, 1977; Entry and Emmingham, 1998; Fang et al., 2010;

Dietrich, 2012; Fedrigo et al., 2014; Schrumpf et al., 2014). In the present work an attempt is made to fill that knowledge gap and measure its effect. This was considered a preliminary step before measuring and comparing ΔSOC between pairs on the boundary of an escaped regeneration burn.

Estimates of ΔSOC accompanying rangeland deforestation and regrowth in Australia have high uncertainty (Henry et al., 2002), which is one reason they were not included in Australia's adoption of the Kyoto Protocol GHG accounting. Similarly, ΔSOC accompanying slower processes such as woody re-colonisation or forest degradation from erosion are not currently included; also as they correspond to 'grazing management', which was excluded from Australia's Kyoto-Protocol accounting. Such fluxes are examined in this study in order to provide estimates of scientific relevance, rather than to be guided by an administrative selection.

Conceptualisation in four dimensions (4D, the three Cartesian axes plus time) has provided scientific insight and management options, in ecology and C dynamics (e.g. Ward, 1989; Dean et al., 2004; Dean and Roxburgh, 2006; Turnbull et al., 2008; Colyan and Ginzburg, 2010). Woody thickening in rangelands, has significant 4D diversity in SOC concentrations (Boutton et al., 1998). In the present work, solutions are investigated for solving earlier dissonance over ΔSOC for significant rangeland processes accompanying LUC, with data from Australia and the USA.

Apart from logging and deforestation, carbon stocks in Australian forests have also been reduced by inadvertent or illegitimate anthropogenic fire. For example, fires prior to the 1940s in Victoria and Tasmania which initiated significant short- and long-term carbon emissions from carbon-dense forests, were of human origin (Blanford and Stratton, 1939; Rankin, 1947 p11) and rural fires generally in Australia are at least 50% anthropogenic in origin (e.g. Forestry Tasmania, 2001; Bryant, 2008). These effects will not be calculated in the present work but nevertheless form a background which must be considered when discussing options for managing LUC.

Rangelands supporting commercial livestock grazing are contested ground in which production of meat and other animal products for the increasing human population,

increasingly competes for space with nature conservation and conserving or replenishing carbon stocks (e.g. Glenn et al., 1993; Schuman et al., 2002; Reid et al., 2004; Dutilly-Diane et al., 2006; Khan and Hanjra, 2009; Janzen, 2011). These rangelands have commonly experienced net vegetation and soil loss (e.g. Allen, 1983; Fanning, 1999; Zucca et al., 2010; Dotterweich, 2013) corresponding to net C emission. Whereas less-degrading management would reduce those emissions, or more specifically, reduced deforestation, protection and enhancement of soil organic carbon (SOC), and reforestation would do so (Henry et al., 2002). Moreover, reversal of land degradation linked to the carbon emission process can theoretically replenish the lost C (Howden et al., 1991; McKeon et al., 1992; Glenn et al., 1993; Walker and Steffen, 1993; Henry et al., 2002). Consequently, rangeland SOC can theoretically act as a C sink in support of greenhouse gas (GHG) mitigation projects (McKeon et al., 1992; Glenn et al., 1993; Walker and Steffen, 1993). The correlation between SOC and plant biomass means that SOC in many rangelands can be influenced by: overgrazing and land rehabilitation; deforestation and regrowth; woody thickening; fire; and climate change (Batjes and Sombroek, 1997; Reeder, 2002; Smith and Johnson, 2004; Luo et al., 2007).

The refilling of that depleted carbon stock (henceforth termed re-sequestration) contrasts with sequestration projects storing C in a form or location different to its origin (e.g. afforestation or power-station carbon capture and storage). The implementation of carbon re-sequestration projects has been limited by uncertainties in their potential magnitude, duration, location and profitability.

Magnitudes of change in soil organic carbon (ΔSOC) have higher uncertainty than associated changes in rangeland biomass (Henry et al., 2002). The difficulty in measuring and forecasting rangeland SOC may have resulted in conflicting reports on the direction and magnitude of ΔSOC that accompanying LUC, especially where there is only tenuous evidence for causation between observed effects and anthropogenic activity. The magnitude and direction of ΔSOC, can differ between decadal and centennial time-scales for rangeland grazing (for example), being more-definitive in the longer term (Piñeiro et al., 2006). Similarly, shallow measurement of SOC (≤0.2 m) can both greatly underestimate the SOC pool and lead to incorrect

conclusions on the direction of ΔSOC following management effects (Harrison et al., 2011). Uncertainty, which is higher for ΔSOC, stifles climate change mitigation projects in the rangelands (Harper et al., 2007; Brown et al., 2010) and it has tacitly prevented inclusion of some rangeland SOC fluxes in national GHG accounts, thereby detracting from accounting accuracy.

1.2 LUC processes to be studied

Terrestrial high carbon stocks in nature occur in four forms: high carbon density, large area of low carbon density, large area of medium to high carbon density, and thick layers of fossilised biomass. Examples of the third are the soil carbon of the arctic tundra and the prehistoric oak forests of Europe, neither of which occur in Australia. This thesis examines land-use effects on C in carbon-dense forests, the 'tall open-forests', focussing on those in Tasmania; and the more-sparse carbon distribution of the expansive Australian commercial rangelands and theory adapted for rangelands overseas. These are two extremes but both have undergone relatively intense commercialisation, and the greenhouse gas emissions of these have been viewed as significant, as have biodiversity effects and economic effects, in different quarters. Other commonalities are the poor level of climate-change-relevant information available for commercial activity in each and some of the points of contention regarding emissions.

1.2.1 Australian tall open-forests

Tall open-forest (TOF) has the tallest stratum >30 m high and with a canopy cover of 30–70% (Specht, 1970). Most of the TOF in Australia is mapped in the National Vegetation Information System (NVIS) as 'eucalypt tall open-forest' (Commonwealth of Australia, 2012) and non-eucalypt-dominated TOFs have been integrated into other categories such as rainforest. The total area of TOF in Australia is 3.6 Mha according to the NVIS dataset or 4.9 Mha in the most recent State of the Forests Report (MIG and NFISC, 2013). Reasons for the discrepancy in area are the

difference in definition of canopy closure for ‘tall open-forest’; 30–70% in Specht (1970) and 50–80% in MIG and NFISC (2013); changes in forest identification; and data updates (MIG and NFISC, 2013, p47). The NVIS dataset includes extant coverage, as of 2012, and the pre-European settlement coverage (i.e. ‘pre-1750’).

The temperate forests of Australia have a long history of Aboriginal burning (>40,000 years; Jackson (2000)) followed in ~1750 by European settlement. Australia’s primary-forests have been subject to competing demands for over a century, from infrastructure to agriculture, forestry and conservation and they are still subject to LUC (Kirkpatrick, 1986; Bradshaw and Ehrlich, 2015). From LUC the TOFs in Australia have lost ~20% in condition and ~20% in area— an overall reduction in biomass of 40% (Mansergh and Cheal, 2007). In much of the residual native forest, direct impact is through timber extraction for fuel, lumber, pulpwood, oils and charcoal; and indirect impacts are through hydrology, and intensification of pests, weeds and pathogens; and increasingly, climate change, and fragmentation (Bradshaw and Ehrlich, 2015). Owing principally to timber harvesting the remnant south-eastern Australian temperate forests are at 60% of their potential carbon stock (Roxburgh et al., 2006b).

In the remnant carbon-dense TOFs the age class structure of dominant stratum has been dramatically changed, especially with the increase in the pulpwood industry in the twentieth century (Kirkpatrick, 1986). Due to logging, forest carbon in the 14.5 Mha of remnant eucalypt forests of south-eastern Australian (including 3.4 Mha of TOF), averaged spatially across different logging intensities, has decreased in the living biomass, litter and SOC pools by approximately: 15%, 10%, and 4% respectively (Roxburgh, 2009, Fig. 8-2). In a logged TOF near the centre of mass of that forest region, the aboveground biomass and litter carbon pools were 39(±5)% (172(±24) Mg ha^{-1}) below capacity (Roxburgh et al., 2006b). Due to the long, limiting half-life of SOC, it is likely to take ~2,000 years to equilibrate under the new biomass regime (Liski et al., 1998; Hibbard et al., 2003), i.e. a ‘long-term’ change.

More indirectly than from logging, reduced growth is forecast for eucalypts in TOFs due to reduced water availability under climate change (Bowman et al., 2014), which constitutes a carbon emission when compared with earlier biomass stocks. Detailed

climate modelling has confirmed an increased fire danger index forecast for Tasmania this century, especially for the southern region which contains the most carbon-dense TOFs and largest trees (Fox-Hughes et al., 2014). The effect of climate change on TOFs will be examined in the present work through modelling.

1.2.1.1 Characteristics of tall open-forests

This section introduces aspects of the ecology, industrial usage and studies to-date, of tall open-forests relevant to improving their carbon accounting, as studied in Chapters 2-to-7. The type of TOF known as 'wet-eucalypt' forest is subdivided into 'wet-sclerophyll' and 'mixed-forest' (Kirkpatrick et al., 1988). Mixed-forest occurs in southeast Australia and is a eucalypt TOF with a rainforest understory. It is ecotonal, occupying a mesic zone between lower-water-balance eucalypt forest and higher-rainfall, lower-fire-frequency rainforest (Gilbert, 1959). Mixed-forest is a form of rainforest (Kirkpatrick and DellaSala, 2011), though it is an overlap of the TOF and traditional rainforest categories. However, in this thesis when rainforest is mentioned it will not refer to mixed-forest, in order to avoid confusion with the present mapping of vegetation types used in Australia. Mixed-forest is common in Tasmania and was previously common in Victoria, (Australia) where it is now rare and termed 'ecotone' forest (Petrie et al., 1929; Fedrigo et al., 2014). Most scientific studies on *E. regnans* forest have been carried out in Victoria, and some in Tasmania, where the larger individual specimens now remain. The Tasmanian island population of *Eucalyptus regnans* F. Muell. (swamp gum) mixed-forest is genetically linked to the Victorian wet-sclerophyll *E. regnans* (mountain ash) population via the Otway Ranges (Nevill et al., 2010). Mixed-forest occurs in NSW and QLD with the eucalypts emergent mostly over sub-tropical rather than temperate rainforest (Gilbert, 1959). Mixed-forest is not present in Western Australia (WA), where the only TOF is wet-sclerophyll forest.

Swamp gum is still a dominant ecological component at over 500 years of age in mixed-forest in Tasmania (Wood et al., 2010), but without sufficient fire to initiate germination and sapling-stage eucalypts in place of the rainforest understorey

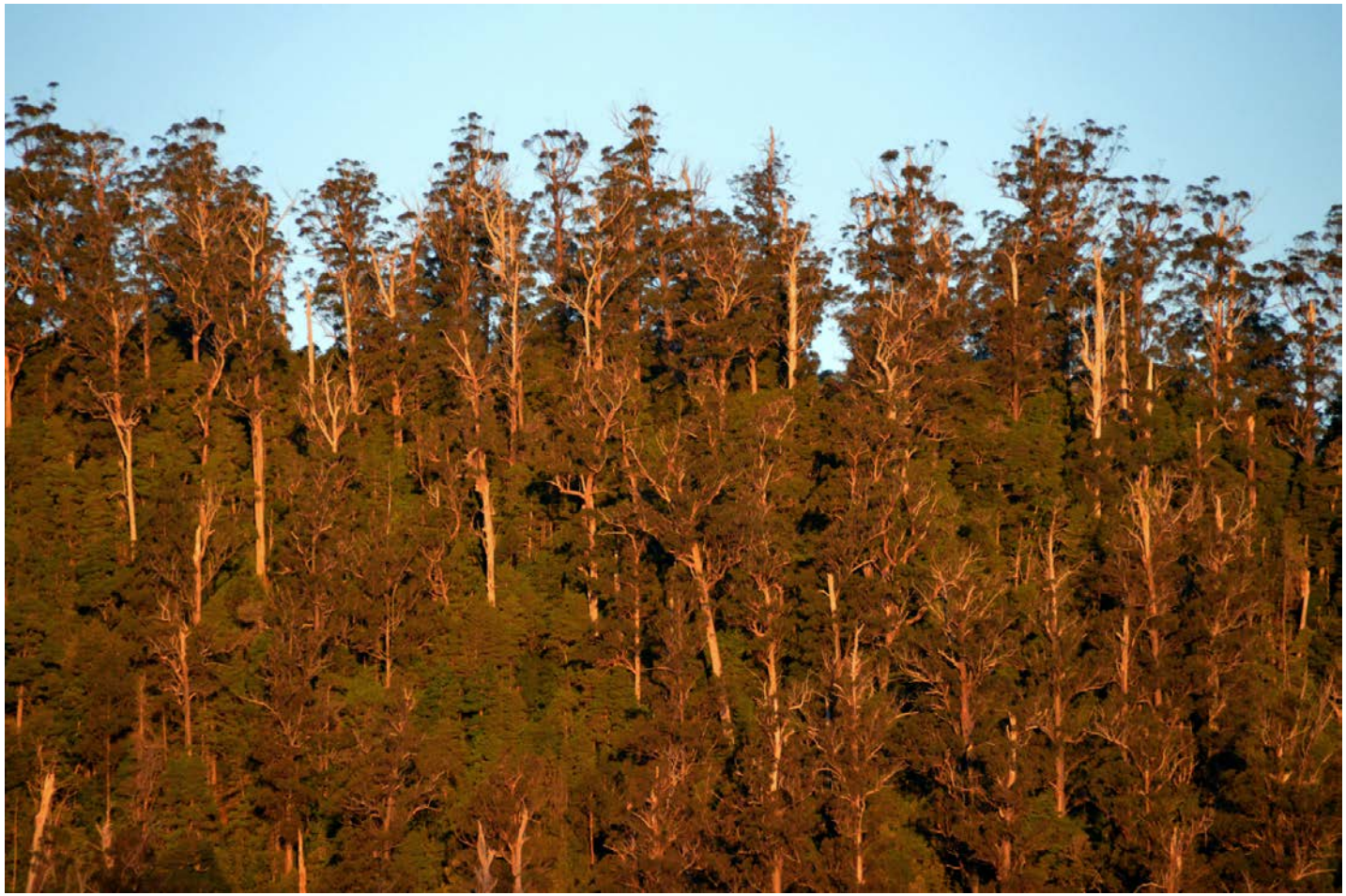


Figure 1-2. Mixed-forest TOF, *E. regnans* (swamp gum) canopy and rainforest understory. The understorey is comprised of myrtle (*Nothofagus cunninghamii* (Hook. f.) Øerst.) and sassafras trees (*Atherosperma moschatum* (Labill.)). North side of upper Styx Valley, depauperate compared with south side, but lightly selectively logged in mid-20th century.

Figure 1-3. Uneven-aged TOF. Uneven-aged eucalypt TOF, following fire. Note [yellow flowering] *Acacia dealbata*, a coloniser after fire. Lower Styx Valley, Tasmania. For scale note truck in foreground and inset showing majority of the cable-logging coupe.

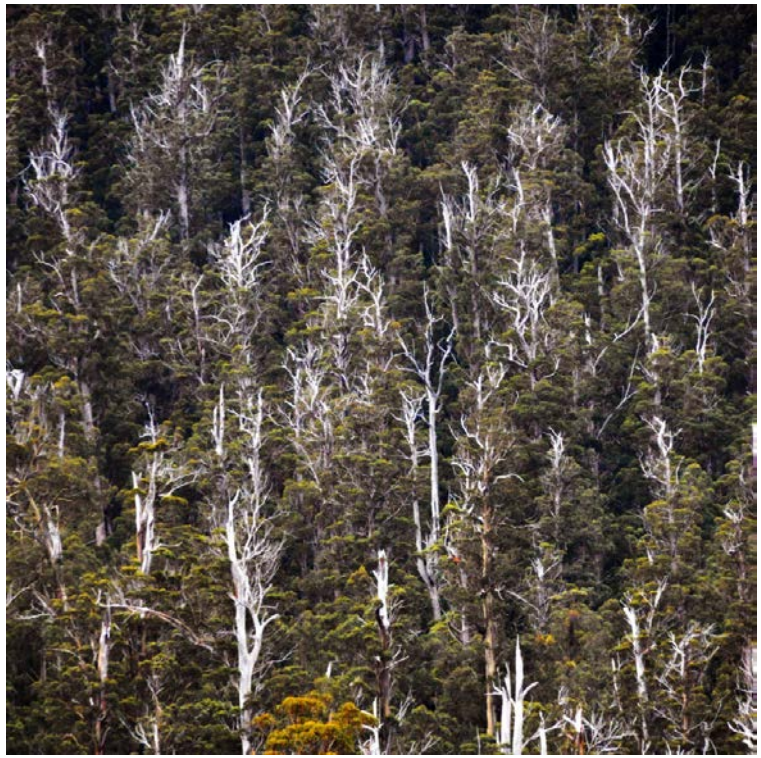


Figure 1-4. TOF partial recovery after fire. The rainforest understorey appears mostly absent, indicating that one or more fires have at least temporarily deleted mature rainforest individuals from the stand, i.e. it is not mixed-forest. Lower north Styx Valley, viewed from across Styx River in coupe SX0019I.

Figure 1-5 Canopy stratification of mixed-forest TOF. Mixed-forest, *Eucalyptus regnans* (~72 m high) above myrtle and sassafras, Edge of logging coupe SX009C, Styx Valley, Tasmania. Eucalypt in foreground, left: DBH= 4.56 m, with original crown and mature myrtle hemi-epiphyte joint up 5.4 m height on the right hand side and sassafras joint up to 4.7 m on left hand side (Styx Valley). (Same tree in Figure 5.7).

(Gilbert, 1959; Cremer, 1960; Ashton, 1981) then the mixed-forest eventually becomes rainforest. Similarly, in the absence of stand-replacing fire, wet-sclerophyll

Figure 1-6 Legacy carbon from the earlier mixed-forest. A *Eucalyptus regnans* log spans a creek centred in a 200 m wide gully mapped as rainforest— typical of the blurred spatio-temporal boundary between TOF and rainforest as seen from a carbon dynamics perspective. Cliff Creek, Styx Valley, Tasmania.

can lead to mixed-forests (Gilbert, 1959; Ashton and Attiwill, 1994). The documented area of rainforest has changed with the definition of rainforest, according to the percentage crown cover of eucalypt remaining within the mixed-forest (Hickey et al., 1993). In addition to the definition-dependent, blurred spatial boundary between the two there is also a blurred temporal boundary, as coarse woody debris and soil carbon, representative of either forest type, persists to a degree depending on its half-life. This material constitutes 'legacy carbon' (Harmon, 2001; Harmon, 2009) (Figure 1-6). Empirical studies attempting to compare soil carbon stocks in rainforest and mixed-forest will thus be futile (e.g. Dietrich, 2012) until the transition between the two is almost complete for all forest attributes. As a first step in rationalising SOC for different forest types or ages, the SOC close to mature trees, where it can be more concentrated, is studied in the present work.

Eucalyptus regnans-dominated forest is amongst the most carbon (C)-dense primary-forest worldwide (Keith et al., 2009; Wood et al., 2010; Fedrigo et al., 2014), mature individuals of *E. regnans* have the highest recorded, gross annual C sequestration rate globally (i.e. not counting C emissions) (Sillett et al., 2015), and *E. regnans* is the tallest living angiosperm: up to 115 m (Ferguson, 1948; Ashton, 1975a; Ashton, 1981; Mace, 1996), of similar maximum height to the tallest gymnosperm *Sequoia sempervirens* (Sillett et al., 2010). *Eucalyptus regnans* is the most mesophilic of the TOF eucalypts (Petrie et al., 1929), with a high affinity for moist, well-drained soils and minimal investment in individual fire-resistance (such as thick bark or lignotubers). This, together with the global link between tree height and water availability (Moles et al., 2009) helps to explain its outstanding height. Stands of tall trees in remnant primary TOF in each of WA (*E. diversicolor* growing up to ~ 87.5 m high (Campbell, 1921) intermingled with *E. jacksonii*), VIC (*E. regnans*), and Tasmania (*E. obliqua*) accommodate elevated tall tree walks for tourists, while most stands service the timber industry.

With particular climate and fire frequency, a rainforest understorey of myrtle trees (*Nothofagus cunninghamii* (Hook. f.) Oerst.) in *E. regnans* mixed-forest can be replaced by young *E. regnans*, thereby increasing the carbon stock once the younger eucalypts reach canopy height (Mackey et al., 2008). That combination possibly explains the high carbon stock observed in some places in the Victorian Central Highlands (VCH) (Keith et al., 2009). However that combination may be more metastable with respect to fire and time than the mixed-forest, possibly requiring more-frequent low intensity fire for maintenance. For example Fedrigo et al. (2014) found marginally higher carbon stocks in mixed-forest than in wet-sclerophyll forests in the VCH. Mixed-forests are often also uneven-aged with respect to the eucalypts (Bowman and Kirkpatrick, 1984; Turner et al., 2009; ANM, c1960), though even-aged stands of *E. regnans* mixed-forest are not uncommon (ANM, c1960). It must be noted that the carbon allometrics (allometric equations) established for a low-fire-frequency location may introduce additional errors if used in a high-fire-frequency location (and vice-versa), due to tree-hollow enlargement with more-frequent or more-intense fires (e.g. Figure 1-8).

Tasmania's TOFs, especially the more-mature *E. regnans* mixed-forests have been prized for pulpwood and lumber, initially for newspaper and then as the major source feeding the hardwood pulpwood market of the Pacific rim, especially Japan, until 2012 (The Mercury, 1941; ANM, 1979; WRI, 2010; WRI, 2014). A portion of primary TOF in Tasmania, rather than being converted to plantation, was converted (and more still continues to be converted) to secondary, more-eucalypt-dominated forests, under a harvesting cycle. Long-term C stock (when including wood-products in the total) is forecast to decrease upon primary forest conversion (Harmon et al., 1990).

Carbon stocks representative of any particular land-use and forest-type, are best measured as a long-term, temporal average (Nabuurs and Schelhaas, 2002; IPCC, 2003). For TOFs the stand-level biomass oscillates in the long-term with natural wildfire (Dean et al., 2003). A considerable portion of killed biomass remains on site as coarse woody debris (CWD) and is not emitted to the atmosphere. While new biomass increases after the fire, the CWD decomposes sooner than if it had remained alive, except possibly for the small portion left as charcoal which may turnover on a centennial time scale (Singh et al., 2012), though charcoal is not long-lived in some microenvironments (Zimmermann et al., 2012). Appropriate modelling is used in the present work to cover long-term effects.

Carbon content varies spatio-temporally within one forest type, linked in part to how sporadic is fire, so spatial averages from small study areas are more likely to be different, for example for *E. regnans* in the VCH: 2.52 ha with 689 Mg ha^{-1} (Fedrigo et al., 2014), 0.729 ha with 706 Mg ha^{-1} (Sillett et al., 2015) and 3.18 ha yielding 1053 Mg ha^{-1} (Keith et al., 2009). Temporal averages are harder to empirically measure than spatial averages, due to relatively short human longevity compared with that of mature trees in primary forests. Comparison of rainforest and TOF carbon stocks, an argument broached by the forestry industry to justify maintenance of eucalypt stock rather than allow succession to mixed-forest or pure rainforest, is problematic for this reason; it is briefly examined in this work (Figure A-IV-1).

Additionally, if considering carbon stocks alone, it can be difficult to differentiate between site index effects, logging history and stand age effects. For example in the

Tasmanian timber production estate, TOF ≥110 years of age have on average 186 Mg ha^{-1} of carbon in AGB, and a maximum of 748 Mg ha^{-1} (Moroni et al., 2010). Australia-wide, it is unfortunate that data so important to climate change modelling, as geographically-defined forest history, timber inventory, plot data, and weigh bills, are held as commercial-in-confidence. The interacting effects of fire and logging influence the landscape mosaic of these forests (Lindenmayer et al. 2009), with spatio-temporal implications for determining C dynamics. Determining stand-level C stocks in forests of varying seral stages is a crucial first step to investigating landscape-level, long-term C dynamics.

Over the last two decades carbon accounting of deforestation and industrial forestry has prompted research into allometrics of large trees in relation to climate change modelling and carbon credits (e.g. Brown, 1997; Dean, 2003; Chave et al., 2005; Dean and Roxburgh, 2006; Nogueira et al., 2006; Ngomanda et al., 2012). Missing information necessary for allometric equations suitable for carbon accounting for Australian timbers (Table 1-1) illustrates the paucity of scientific research into mature trees, with attention historically focused on young trees of the secondary forests, where financial investment is higher. Modelling carbon dynamics relevant to climate modelling requires parameterization of allometrics for the full range of tree sizes and major (in terms of biomass) forest components. Accompanying forest usage and clearance, the size of the largest trees remaining in existence has been decreasing (Herrmann, 2006; Lindenmayer et al., 2012). This makes the information gap smaller for current forests, but makes accounting for the earlier primary forests and our effect on them, and accounting for potential future forests, more difficult. The missing allometrics means that effects of earlier forest usage cannot accurately be integrated into climate change modelling. The reader is referred to Melson et al. (2011) and Henry et al. (2015) for a comparison of error margins between different types of allometric equations and recommendations for improvement in allometrics suitable for carbon accounting globally. In the present work judicious use of proxy allometrics is explained and those for contributory parts of trees are derived for improvement in the allometrics of large trees of *E. regnans*.

Table 1-1 – Maximum DBH used for calibrating publicly accessible, species-specific carbon allometrics, compared with maximum recorded tree diameters for some common TOF canopy species. DBH is indicative of gross sequestered carbon over lifetime, and of flutes in buttress. Allometrics are generally available only for trees up to around half of maximum size.

State habitat	TOF Species	Maximum DBH contributed to allometric (m)	Maximum DBH recorded (m)
TAS/VIC	*E. regnans*	6.45 (Dean and Roxburgh, 2006)	10.76 (Ashton, 1975a)
VIC/TAS	*E. obliqua* (messmate/stringybark)	3.50 (Keith et al., 2000)	6.53 (McIntosh, 2015) “Mt Cripps” TAS 6.31 (Maiden, 1904) VIC
WA	*E. jacksonii* (red tingle)	None available	5.92 (McIntosh, 2015) “Pleated Lady”
VIC/TAS/NSW	*E. ovata syn E. gunnii var. acervula* (black gum)	0.3 (Paul et al., 2013)	5.82 (Maiden, 1904)
VIC/TAS/NSW	*E. delegatensis* (gum-top-stringybark/alpine ash/white-top)	0.83 (Keith et al., 2000)	5.73 (McIntosh, 2015) “Troll” TAS
WA	*E. diversicolor* (karri)	1.51 (Grierson et al., 2000)	4.14 (McIntosh, 2015) “Hawke”
NSW/QLD	*E. pilularis* (blackbutt)	1.29 (Applegate, 1982)	4.07 (Maiden, 1917)
NSW/TAS/VIC	*E. viminalis* (manna gum)	0.298 (Clifford et al., 2013)	3.50 (McIntosh, 2015) “White Knight”
WA	*Corymbia calophylla* (marri)	1.10 (Grierson et al., 2000)	3.44 (McIntosh, 2015) “Poole”
WA	*E. marginata* (jarrah)	1.84 (Grierson et al., 2000)	3.28 (McIntosh, 2015) “Hadfield”
NSW/QLD	*Syncarpia glomulifera* (turpentine)	0.70 (Eamus et al., 2000)	2.91 (Maiden, 1917)
NSW/QLD	*E. microcorys* (tallow-wood)	0.90 (Eamus et al., 2000)	2.72 (Maiden, 1917)

1.2.2 Australian commercial rangelands

The Australian commercial rangelands are essentially dedicated to the export market of livestock products, whereas prior to European settlement (1788) they were used by subsistence-based societies. The accompanying changes in the native forests in Australia's rangelands have principally been through degradation, timber extraction, deforestation, regrowth and woody-thickening. The definition for rangeland that is adopted here is areas where domestic livestock 'rove at large' (Chambers, 1908) in natural or semi-natural vegetation inhospitable to arable agriculture— a subset of the 661 Mha rangeland zone of Donohue et al. (2005). Excluding reserves and mesic arable land from that zone, leaves the land which is used largely for the generation of profit from grazing domestic livestock (henceforth commercial rangeland).

The Australian commercial rangelands, differ from those elsewhere (e.g. the grasslands of South America and the Great Plains of the USA), in that they are predominantly shrubland, woodland and forest with lesser amounts of scrub, heath and herbland (Carnahan, 1977; Luly, 1993). Nevertheless, Australia and the USA share similarities in the industrialisation of their arid and semiarid rangelands (e.g. investment, speculation, social mores and exports) (Heathcote, 1969). There are also parallels between the historical livestock-driven environmental degradation processes in Australia, the Americas and southern Africa (Pickup, 1998).

In the present work the Australian Government's definition of forest (DCCEE, 2010) is used: a stand of trees covering at least 0.2 ha, attaining at least 2 m high at maturity and with at least 20% projected canopy cover. This definition extends the concept of forest to vegetation previously considered in Australia to be woodland, scrub, or shrubland (Specht, 1972). The height of 2 m is low compared with that typical of temperate forests but the basic density of several major Australian semiarid species is comparatively high, near 1000 kg m^{-3} (Ilic et al., 2000). A projected canopy cover of 20% corresponds to approximately 11% foliage projected cover (FPC) — a threshold used in remote-sensing analyses of rangelands to determine which image pixels represent forest cover (Scarth et al., 2008).

For many rangelands, woody vegetation plays a major ecological role, through vegetation types such as: (a) shrubland, woodland, open-forest, riparian forests in grasslands, or savannah; (b) improved pastures after deforestation; or (c) extant grassland experiencing woody thickening; (e.g. Carnahan, 1977; Lund, 2007; Ellis, 2011). The term 'wooded-' rather than 'forested-' is used here with respect to rangelands, to avoid confusion between different countries' definitions of forests, and to include the woody-thickened state.

(a)

(b)

Figure 1-7 – Change in carbon with rangeland overgrazing. Northern Territory, Australia. Stocking levels of livestock are often kept high in the hope that plentiful rains will produce sufficient feed. Initially, native vegetation is lost and topsoil lost (with reduced grass and woody cover) is lost **(a)**, then less-palatable species proliferate including 'woody-weeds', especially near water sources where grazing has been more intense (**(b)**, same paddock as in (a)). One indication of overgrazing is raised livestock fatalities through starvation and sometimes from eating poisonous vegetation (a). Regions of both situations can be detected by remote-sensing.

Australian rangelands were studied to determine their contribution to anthropogenic climate change, such as the C flux, major contributory pools and potential for C re-sequestration, and the likely effect of further climate change. In recent decades the highest rates of deforestation in Australia were in the State of Queensland (QLD). Changed forest use in QLD offers the highest, net sequestration in the rangelands nationally, through reduced deforestation and managed regeneration, but also with biodiversity plantings and plantations (Bray and Golden, 2008; Fensham and Guymer, 2009). Both QLD and the State of New South Wales (NSW) are studied in finer detail, facilitated by the unique remote-sensing and vegetation data available for them, with a view to determining re-sequestration rates and optimal locations for re-sequestration. For NSW, this extended to an evaluation of finances; and an understanding of the relevance of the findings for NSW to rangeland elsewhere is facilitated by a global climate and biome comparison.

1.3 Choice of methods deployed

Methodology was a dominant part of this project in that much of it, together with conceptualisation, facilitated advances. Six main methods were deployed throughout the project and the following overview provides an integrated perspective.

1.3.1 Standard: forest mensuration

The standard forest mensuration technique used in the TOF was principally DBH measurement (tree diameter at breast height, i.e. 1.3 m (standard Australian, European and British height)) from the ground on the high side of the tree, and including adjustment for stem slope). For larger trees horizontal-ness of the girth tape was checked with a clinometer. DBH measurement was used throughout as it is relatively straightforward and has the benefit within the Australian system of universality in databases and allometric equations for biomass or carbon. Some tree allometrics include height, traditionally used by the timber industry as it helps indicate log lengths for milling. However it is rarely possible to measure height while

stood on the ground under a closed understorey canopy, such as in mixed-forests. A clinometer and laser rangefinder were used for measuring heights of trees of specific interest. Incorporating tree height into the allometric is less informative for carbon content or even distortional when combined in an allometric with DBH once a tree matures, as there would be different parameters in the allometric at different stages of a tree's life, for example after crown loss (e.g. Biggs, 1991) or crown regeneration. One must also be cautious if applying allometrics developed from a specific ecological category of trees to trees outside that category, even if DBH is the only equation's variable, for example non-senesced trees (e.g. Sillett et al., 2015), or trees from a location with a different fire frequency (Figure 1-8) as fires can increase tree hollow size (Adkins, 2006). There are some downsides to using DBH, for example internationally there are different heights at which DBH is measured, notably the Americas (1.33 m, Gough et al. (2007a); 1.36 m , Nogueira et al. (2006); 1.37 m, Sillett et al. (2015); 1.4 m , Amichev et al. (2008)), New Zealand (1.4 m, Beets et al. (2012)) and Japan (1.2 m, Kitahara et al., 2010). Both within Australia

Figure 1-8. Crowns of living *Eucalyptus jacksonii* (Red Tingle) trees, TOF ~75 years after intense fire. Now senescent, with crown loss and regrowth. Allometrics based on DBH may not represent carbon loss from the large hollows and loss of previous crown, if developed using trees from a location without the earlier intense fire. Drawn from photos supplied by Assoc. Prof. Grant Wardell-Johnson.

and overseas, for ecosystems with smaller trees, such as those in semiarid places with multiple or low-branching stems, diameter measurement at 0.5 m aboveground or lower, is common (e.g. Burrows et al., 2000; González-Roglich et al., 2014). Fortunately, in Australian temperate eucalypt forests, 1.3 m has been used almost universally and is common globally (IPCC, 2003, p4.101).

Trunk diameter measurement using a tape measure is actually a measure of circumference and the conversion to diameter assumes that the trunk is circular. A circle is the smallest area of any perimeter measurement, and any stem non-circularity or flutes (gaps) between spurs on the buttress gives a falsely high reading of area, with the difference between that and a circle being the 'cross-sectional area-deficit'. For buttressed trees, such as *E. regnans*, in place of the standard DBH measurement Sillett et al. (2015) formulated a 'functional diameter' which was a measure of the wood cross-sectional area without the area-deficit. The functional diameter potentially allows higher consistency in formulating biomass allometrics, but it is still necessary to measure the standard DBH measurement as the starting point. The present work uses the standard DBH measurement and investigates parameterisation of the cross-section, including the cross-sectional area-deficit.

1.3.2 Citizen science

Teams of field volunteers were trained by volunteer scientists (some who had been employed in professional carbon accounting) and deployed for stand-level measurements of DBH and CWD in the TOFs. The deployment of a large number of non-professional data collectors is a sledge-hammer technique to obtain large data sets, which is increasing in popularity and known as 'citizen science' (Bonney et al., 2014). Error margins may be higher from using non-professional scientists to collect most data points but this is shown to depend more on, for example, the quality of the training, survey protocols and appropriate usage of statistics (Lewandowski and Specht, 2015). Other than during training, several quality control methods were used. For example, after transcription to digital form, data outliers in volume or mass calculations, were rechecked to whatever degree necessary, including re-

measurement. Systematic errors that can accrue when only one person, with a slightly incorrect method, collects all the data points, would most likely be attenuated in citizen science.

1.3.3 Opportunistic: soil carbon augering in logged forests/stumps

The most theoretically sound method to determine total SOC for a locality is to measure the whole soil profile as a single unit. This can be done either by analysing all the soil dug from a pit, or by extracting a representative number of entire profile cores then measuring their carbon content. These methods were impractical for remote Tasmanian forests as the necessary machinery could not be transported over rugged, mountainous terrain whilst leaving the forest intact. Soil samples were collected in Tasmanian TOFs, primarily with hand auger equipment: an Eijkelkamp© soil corer (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands) with the sample contained in the bulk density head. This had the advantage of allowing both bulk density and carbon density to be determined from the same sample. Forestry activity allowed soil sampling directly underneath the trunks of what had until recently been large, healthy mature eucalypt trees, because either their logging provided access to different-sized, central trunk hollows (from which one could auger to the soil below), or some mature trees had simply been pushed over during logging or burnt during logging and later fell over. The trunk hollows in some trees were narrow and tall and allowed only one core whereas others were wide and allowed both a pit and multiple cores. Trees killed by logging also allowed more destructive soil sampling without damaging the tree or its epiphytes, such as when digging larger pits. Two live, large, mature eucalypt trees, the ‘Cave Tree’ and ‘Chapel Tree’, had trunk hollows that could be entered on foot, through large basal fissures (‘walk-in trees’) allowing augering and pits directly underneath the trunk.

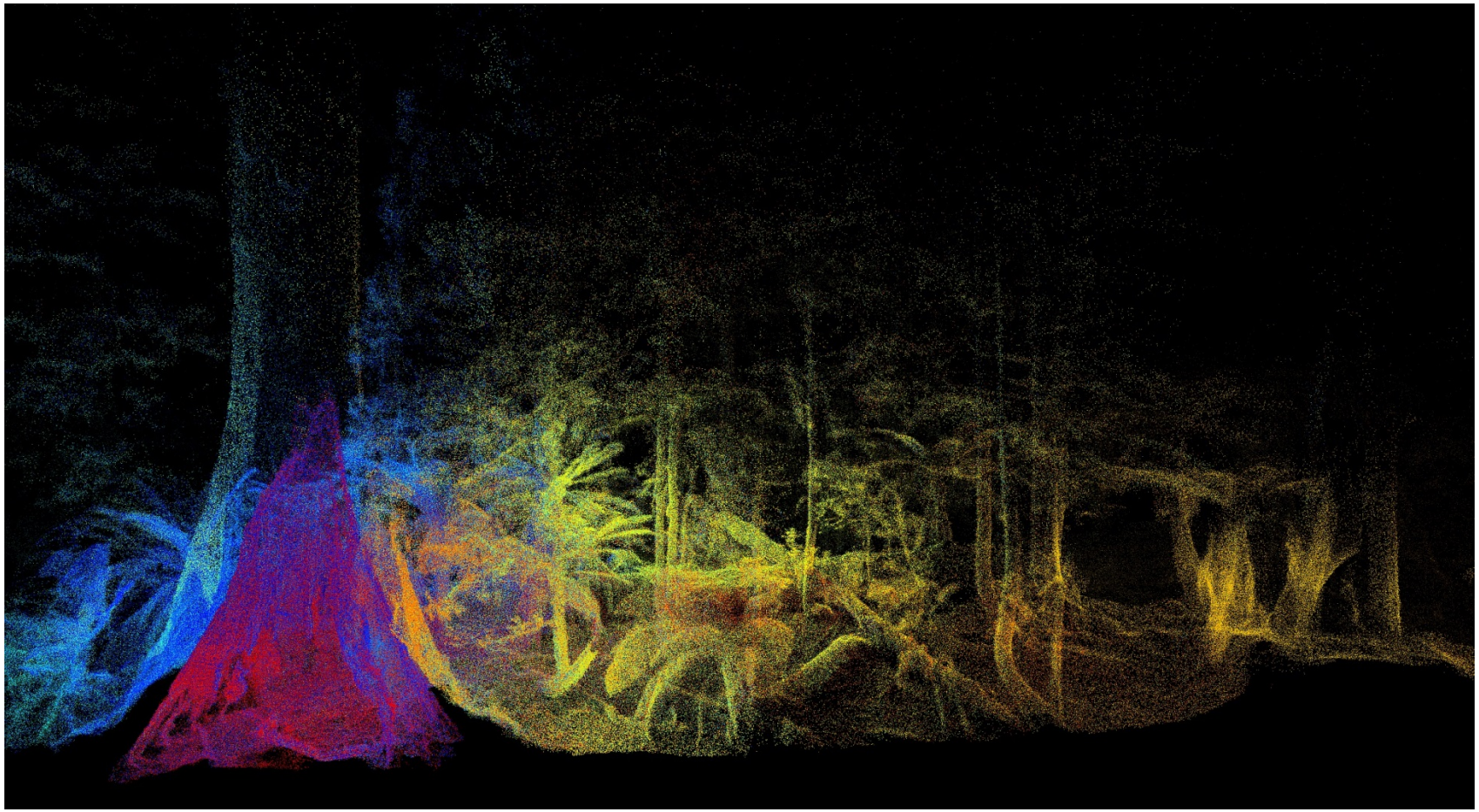


Figure 1-9. Base of 'Cave Tree', an *E. regnans* in the Styx Valley, DBH= 5.15 m, with a 'walk-in' hollow (red dots) in the buttress region; and surrounding understorey and CWD. Data collected by the author with Zebedee, a mobile, terrestrial LiDAR. Picture shows two data collections co-aligned by Robert Zlot and Michael Bosse of CSIRO. Displayed using Meshlab (front clipping plane adjusted to allow median section through hollow buttress). D-SLR photography and photogrammetry with Photoscan, were also performed for this tree. Soil data was collected inside the hollow and analysed for SOC.

1.3.4 New technology: Photoscan, cameras, attached GPS

Terrestrial LiDAR has been used to derive geometrically accurate representations of the surfaces of tree trunks (e.g. Dassot et al., 2012; Bremer et al., 2013) and roots (Teobaldelli et al., 2007; Gärtner et al., 2009; Wagner et al., 2011). Solid objects can be reconstructed from photographs taken with digital cameras by using photogrammetric techniques, particularly the structure-from-motion multi-view-stereopsis algorithms employed by software such as Photoscan (Agisoft, 2015). Digital cameras and Photoscan were used in the present work.

Photography is potentially more suited to the rugged terrain, unstable debris heaps, and the sometimes dense understoreys than is LiDAR, including hand-held mobile LiDAR (Bosse et al., 2012). The aptitude of Photoscan for photogrammetric image processing at the scales and resolutions necessary for mathematically characterising trees has been proven (e.g. Verhoeven, 2011; Morgenroth and Gomez, 2013; Bauwens, 2014; Jiroušek et al., 2014; Liang et al., 2014). The process has high metric accuracy and precision (depending on the geometry and quality of the original photography) and is suitable for trees, tree parts and ground topography measurement. Photoscan uses a self-calibrating photogrammetric bundle adjustment to solve for the camera geometry (camera calibration, including focal length and lens distortions) and to compute camera location and orientation data for each camera exposure station (Kraus, 2007; Verhoeven, 2011; Luhmann et al., 2016). It then uses advanced multi-station image matching techniques to derive a 3D dense point cloud of points common to multiple photographs, from which it constructs a 3D model of the object's surface. Photographs can be acquired using a 'normal' geometry with camera [principal] axes approximately parallel for all camera locations, or with a convergent geometry with camera axes converging towards the object to be mapped (Kraus, 2007). Convergent geometry provides a significantly stronger geometric solution, leading to a more accurate camera calibration and stronger ray intersections. Consequently a convergent camera geometry was adopted in the present work, and because it provided a better photographic coverage of [for example] tree buttress regions. However, it was combined with 'normal' geometry, in accordance with the

Photoscan instructions (Agisoft, 2015), in order to map the diversity of surfaces, such as [long] fallen trees and the cavernous interiors of hollow trunks. The 'normal' geometry method was used alone on long, planar soil profiles such as road cuttings.

The camera most frequently used in the present work had vibration reduction for image stabilization, a combined wide angle and telephoto lens to save changing lenses, and a mountable GPS was custom made to the camera model and which placed coordinates in the image headers and kept a secondary record of the trek suitable for uploading in a GIS. All this new technology was imperative to the design and success of the project; for use in rugged terrain; often in dark, damp conditions; and when speed of traversal and data collection was important.

1.3.5 Remote-Sensing & GIS

Remote-sensing and GIS were used throughout the project, both in TOF and rangelands but at different scales. In the TOF the remote-sensing data was collected through the close-range photography and photogrammetric processing for single trees. For studying rangeland pre-acquired LANDSAT and L-band synthetic aperture radar (PALSAR) imagery, pre-processed into biomass density, were used in land degradation and carbon re-sequestration studies. Google Earth© imagery and pre-acquired aerial photography were used to select study sites with particular characteristics in TOFs and rangelands, as a navigation aid and for identifying specific trees pre-logging, and stumps post-logging. GIS, namely ArcGIS (from ESRI) was used extensively, for navigation, spatial data preparation, combining data layers, cartography, and spatial calculations.

1.3.6 Mathematical modelling, simulation or forecasting

Simulation, modelling, forecasting— three names used in different sciences or computer software fields for a common theoretical activity— seeing what will happen under a set of mathematically linked procedures where one or more variables are mobilised. Tentatively, 'simulation' may refer more to solid objects, 'modelling' to biology, and 'forecasting' more to theorisations involving time as the freed

variable. Modelling was performed for carbon dynamics in both TOF and rangeland but more comprehensively for TOF— landscape-level full life-cycle analysis, whereas for rangeland only individual components were modelled at any one time, e.g. tree biomass or SOC.

The carbon dynamics for TOF was modelled using computer programmes CAR4D (Dean et al., 2003; Dean et al., 2004; Dean and Roxburgh, 2006) and FullCAM (Richards and Brack, 2004; Brack et al., 2006; Waterworth and Richards, 2008). The models were used as published except that the user interface of CAR4D was enhanced in the present work to allow run-time choice for the half-lives of SOC and wood-products in place of the previously hard-coded default values. The designs and formulas for these models have been described previously: CAR4D— Dean and Roxburgh (2006), Dean et al. (2003), Dean et al. (2004), with more detail in Appendix III; and FullCAM (version 3.13.8)—Brack et al. (2006), Waterworth and Richards (2008). The two models have different capabilities:

1) CAR4D

a) designed and calibrated using comprehensive modelling and data for even-aged *E. regnans*-dominated forests,
b) models forest succession of uneven-aged rainforest through a senescing eucalypt canopy,
c) simulates ΔSOC due directly to wildfire, and fire or mechanical disturbance during logging
d) has two SOC pools, one with a default half-life of 2 years and the other 550 years (the slow pool), and
e) simulates C fluxes for two wood-product pools with mill residue included: sawlog and pulpwood.

2) FullCAM

a) models even- and uneven-aged (i.e. including fire effects), mixed-species forests and single-species plantations,

b) does not model forest succession or senescence,
c) aboveground C in biomass limit of 382 Mg ha^{-1} (below peak of *E. regnans* mixed-forest)
d) has three SOC pools but does not simulate ΔSOC due directly to fire or mechanical disturbance during logging
e) does not model ΔSOC (or SOC) below 0.3 m, and
f) simulates C fluxes for several wood-product pools (e.g. sawlog, pulpwood, fibreboard) and one sawmill residue pool.

Both models simulate C fluxes for biomass, necromass, SOC and wood-products, and dissolved organic carbon (DOC) is assumed to remain on-site, in the SOC pool. In CAR4D the sawlog and pulpwood pools include merchantable products and mill residues. Consequently the wood-product half-lives in CAR4D include the half-life of the mill residues. Whereas, in FullCAM sawmill residues (but not pulpwood residues) have a separate C pool with its own half-life. The half-life of sawmill residue is much shorter than that of sawn timber. Therefore: (a) a sawlog half-life of 100 years (for example) in CAR4D denotes greater longevity for the sawlog leaving the forest gate than does a sawlog half-life of 100 years in FullCAM, and (b) in FullCAM, when varying the wood-product half-lives, the half-life of sawmill residue was similarly varied. Thus, the default half-lives in CAR4D for both paper and sawn timber were longer than those in FullCAM.

The default half-lives used for wood-products embodied in the models were derived from contemporary literature. Default half-lives in CAR4D for pulpwood and sawlog were 2 and 40 years respectively. Default half-lives in FullCAM for pulp and paper, fibreboard, construction wood, and mill residue were 1.73, 9.55, 34.3 and 0.2 years respectively. Note that one reason for the low half-life of pulpwood in both models was losses in the paper mill, but that varies greatly between processing types (Richards et al., 2007).

The default half-lives used for wood-products are a combination of the 'in-use' half-lives and decomposition to gas half-lives, tracing the many transformations from forest to gas. Calculating the half-life of Tasmanian wet-eucalypt wood-products entails pursuing a life-cycle-analysis through global freight with ship travel in both

directions, mills, landfills and residue deposition, with high variation in disposal partitioning between countries (e.g. Dias et al., 2007). The carbon conversion efficiency of mill types varies, e.g.: 45–50% for kraft (sulphate) and sulphite, and ~95% for mechanical (but with higher electricity usage) (Dudley et al., 1995). The distribution of mill-types in Japan (the main purchaser of Tasmanian chips) for 1999–2009 was not traced in the present work but kraft pulp mills were commonly used for eucalypt pulp (Greaves et al., 1979) and were a significant component in Japan (McLeod et al., 1995).

Rangeland carbon dynamics were modelled in Microsoft Excel© (hereafter termed Excel).

1.4 Thesis structure in relation to aims

The aim of the project is to improve knowledge of the anthropogenic movement of carbon (C flux) in four dimensions (i.e. three Cartesian axes plus time, 4D) for woody ecosystems, that affects in the first instance, the global climate, with a view towards facilitating mitigation of negative environmental effects. Some example pathways of carbon movement that will be studied are: ecosystem (resource) extraction (in tall open-forest), carbon conservation (in tall open-forest and rangeland), and partial recovery of ecosystem carbon (in rangeland). Only specific examples of these will be studied as neither all industries nor all aspects can be illuminated within the study period. Mathematical relationships linking ecosystem attributes will be developed where necessary to improve carbon dynamics modelling and its incorporation of field data. Theory will be advanced where necessary, to reveal causes or trends in carbon dynamics. The results will be presented in a format that reveals feasible, clear, remedial options for anthropogenic climate change with which the public can engage. Henry et al. (2002) state: 'The challenge for scientists providing policy support and for government agencies dealing with natural resource management is to ensure ecological and economic sustainability in management of landscapes, often against a background of scientific uncertainty.' The present work

attempts to provide knowledge of carbon dynamics to permit better management of carbon and remnant ecosystems.

The thesis has two main sections, the first chapters are concerned with the carbon-dense, timber production forests and the latter chapters are concerned with the sparser forests of the commercial rangelands. The six chapters following the introduction gradually introduce carbon dynamics of the forests, expanding on concepts in the introduction with increasing detail, along the avenues of ecology; mathematics of the trees (allometrics), forest stands and forest soil; contribution of LUC to changes in forest carbon stock and to climate change; and feedback from anthropogenic climate change to the forests. The chapters attempt to follow the highest impact of LUC, by focussing on the most persistent and intensely targeted logging, the impacts of which are tallied. The overall aims of these six chapters are to contribute towards delineating the most influential contribution from that type of LUC upon climate change, and quantifying the simultaneous change in the forest carbon itself. Two of these chapters attempt to add necessary accuracy to SOC stock measurement, which in-turn should carry over to improved quantification of SOC change with LUC. The seventh chapter deals with controversy around the fate of SOC under logging, an existing controversy in the literature and arising from studying the forests. This is necessary in order to provide clarity and hopefully with it, progress in climate change mitigation.

The rangeland section begins with providing theoretical background to solving some controversies in rangeland carbon accounting related to the relationship between vegetation cover and SOC. The background also forms a foundation for the ensuing rangeland chapters, so that the concepts do not have to be frequently reiterated. Estimates of emissions from Australian rangelands have been made since the early 1990s but precision and comprehensiveness of tallying emissions has not improved greatly. Chapter 9 attempts to solve that, to suggest amelioration needs, and to quantify examples of C re-sequestration. Chapter 10 adds further detail, including finances to show plausibility. The discussion chapter (11) explains overarching themes and findings drawn from the preceding chapters. The appendices include

reference topics closely related to the preceding chapters, necessary for completeness.

The opportunities during fieldwork undertaken in this project were unique and with the Tasmanian forests being so little-studied and remote from the general Australian public, several novel items were noted that were scientifically noteworthy but only indirectly related to the major C pools in carbon dynamics. Some of these are written into the Appendices.

Chapter 2 Primary forests, carbon and climate change: functions and management for two hotspots in temperate Australia

This chapter is adapted from a published paper: Dean and Wardell-Johnson (2010).

2.1 Abstract

The prognosis and utility under climate change are presented for two primary TOFs in the temperate zone of Australia, from ecological and carbon accounting perspectives. The TOFs of south-western Australia (SWA) are within Australia's global biodiversity hotspot. The forest management and timber usage from the carbon-dense primary TOFs of Tasmania have a high carbon efflux, rendering it a carbon hotspot. Under climate change, the warmer, drier climate in both areas will decrease carbon stocks both directly and indirectly, through changes towards drier forest types and through positive feedback to climate change. Near 2100, climate change will have initiated the process of a decrease soil organic carbon (SOC), totalling ~30% for SWA and at least 2% for Tasmania. The emissions from the next 20 years of logging primary TOF in TAS, and conversion to harvesting cycles, will conservatively reach 66(±33) Mg of CO_2-equivalents in the long-term– bolstering greenhouse gas emissions. Similar emissions will arise from rainforest SOC in Tasmania due to climate change. Careful management of primary TOFs in these two hotspots, to help reduce carbon emissions and change in biodiversity, entails adopting approaches to forest, wood product and fire management which conserve primary forest characteristics in forest stands. Plantation forestry on long-cleared land and well-targeted prescribed burning supplement effective carbon management.

2.2 Introduction

A new utility of primary-forests is recognised under potentially serious climate change: they retain large reservoirs of C in solid and liquid phases, away from the atmosphere. In neotropical primary-forest there has been a recent increase in that store, thereby offering some mitigation of ongoing climate change (Phillips et al., 2002). The high C stock of primary-forests and high emissions accompanying disturbance are sufficiently significant to suggest that they be recognised when devolving C credits (Luyssaert et al., 2008). Additionally, oldgrowth forests and late-mature primary-forests have lower water demands than regenerating forests—providing lower water stress on surrounding vegetation; and they retain more moisture (and thus refugia) than other stand types (Vertessy et al., 1996; Cooper-Ellis, 1998; Licata et al., 2008; D'Amato et al., 2009; Olson et al., 2012; Ray et al., 2015).

The temperate forests of southern Australia that remain today include, *inter alia*: TOFs, and closed rainforests. Australian southern primary TOFs have an aboveground plant biomass amongst the highest globally (Williams and Brooker, 1997) and some of their eucalypt species are amongst the tallest trees. Tasmania (TAS) and south-western Australia (SWA) are in Australia's temperate south (Fig. 2-1) and produce the tallest and highest-biomass eucalypt stands nationwide (Cremer 1960). Thus such primary forests have C stocks relevant to climate change mitigation, and in turn, release of that carbon can heighten climate change.

For SWA and TAS, only ~37% and ~19% (respectively) of the pre-European extent of TOFs remain (DAFF, 1997). Following industrial activity the proportion of these forests that are primary has been reduced, occupying only ~10% of Australia's forest estate (MPIGA, 2008). With the limited knowledge about such forests, and such vestigial areas, careful management of the identified portions is prudent. Further change in primary forest area and condition are expected to accrue from climate change (Williams et al., 2009).

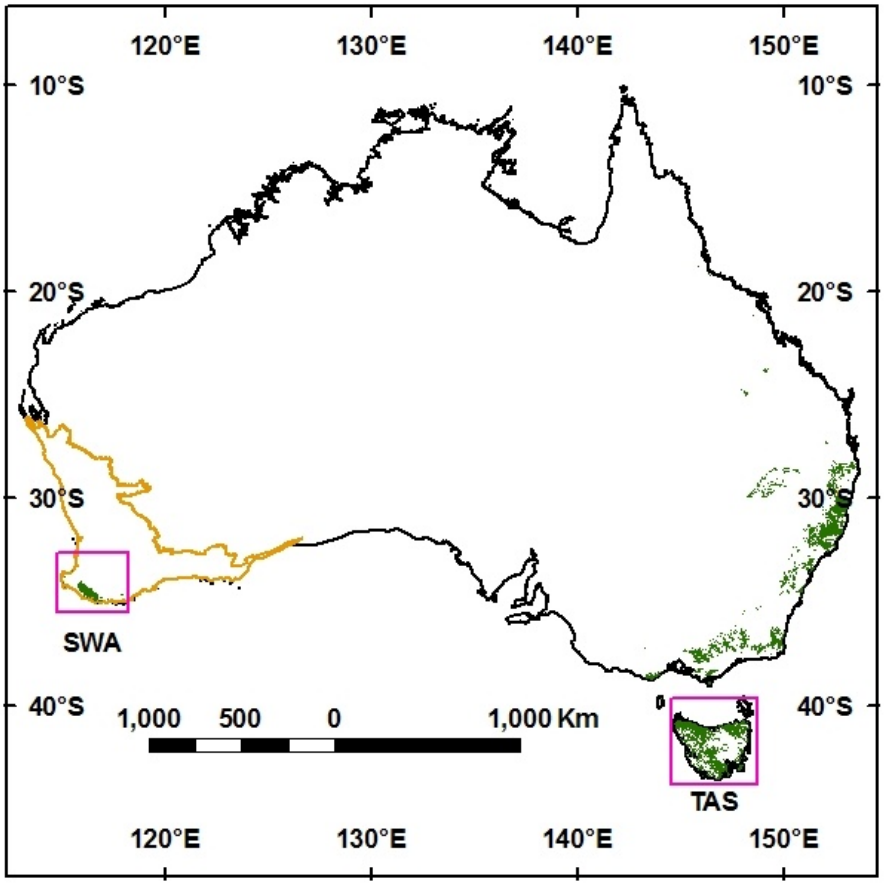


Figure 2-1 Study areas and distribution of pre-European eucalypt TOF in Australia (green). Boundaries of the study areas (pink boxes) for the south-western Australian Floristic Region (yellow) (SWA) and Tasmania (TAS).

The SWA site is within the Southwest Australian Floristic Region (SWAFR, Fig. 2)— one of 34 declared global biodiversity hotspots (Myers et al., 2000). Hotspots can also occur in the C cycle (e.g. Nabuurs et al., 2008). Tasmania (Fig. 2-3) is designated here as such a 'carbon hotspot', due to high C stocks in a small area, it being a national timber-industry focus, and the high level of C emissions accompanying that LUC (Harmon and Marks, 2002; Dean et al., 2003; ABARES, 2010, Chapter 4). In south-eastern Australia the C in live biomass of typical oldgrowth and late-mature primary TOFs is 760–953 Mg ha^{-1} (Coops, 2002; Dean and Roxburgh, 2006)— comparable with oldgrowth coastal Douglas fir (Van Pelt et al., 2004). A C concentration of 1,819 Mg ha^{-1} was also measured (Keith et al., 2009) but its spatial representativeness was undetermined. Tasmania annually produces

~60% of Australia's pulpwood sourced from public native forests, but from only 11% of the area of such forests in Australia, and from only 2.1% of all native forests nationwide. Pulpwood has amongst the shortest half-life of possible wood products. Approximately 35% of the Tasmania sawlog quota (of 300,000 m^3 $year^{-1}$) (the auxiliary product to pulpwood) from public land harvested over the next 20 years will be from ~0.1 Mha of primary-forest (Forestry Tasmania, 2005; Forestry Tasmania, 2008). This allocation was based partly on the principle that it entailed no net C emissions— a maxim noted as unlikely and in need of research even when it was declared (DAFF, 1997). Preliminary analyses (Dean et al., 2003; Dean and Roxburgh, 2006) indicated substantial net emissions for transition of primary-forest *E. regnans* forests to harvesting cycles.

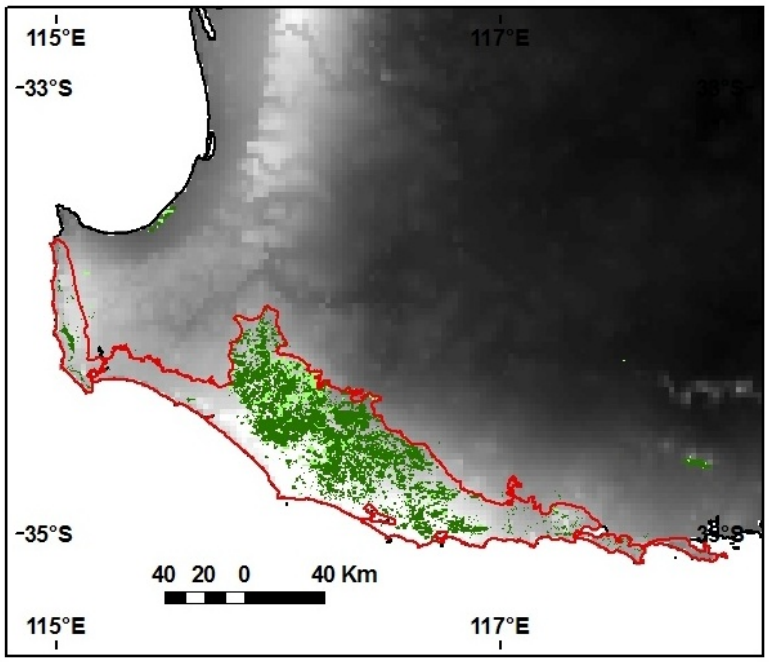


Figure 2-2. Distribution of TOF in south-western Australia. Present— dark green overlaying pre-European extent— light-green (Commonwealth of Australia, 2012). The Warren Bioregion (832,000 ha), which includes almost all of the TOFs in SWA, is outlined in red. Background, grey-level shading is the mean annual rainfall, with higher rainfall shaded lighter (Australian Bureau of Meteorology). Geographic projection.

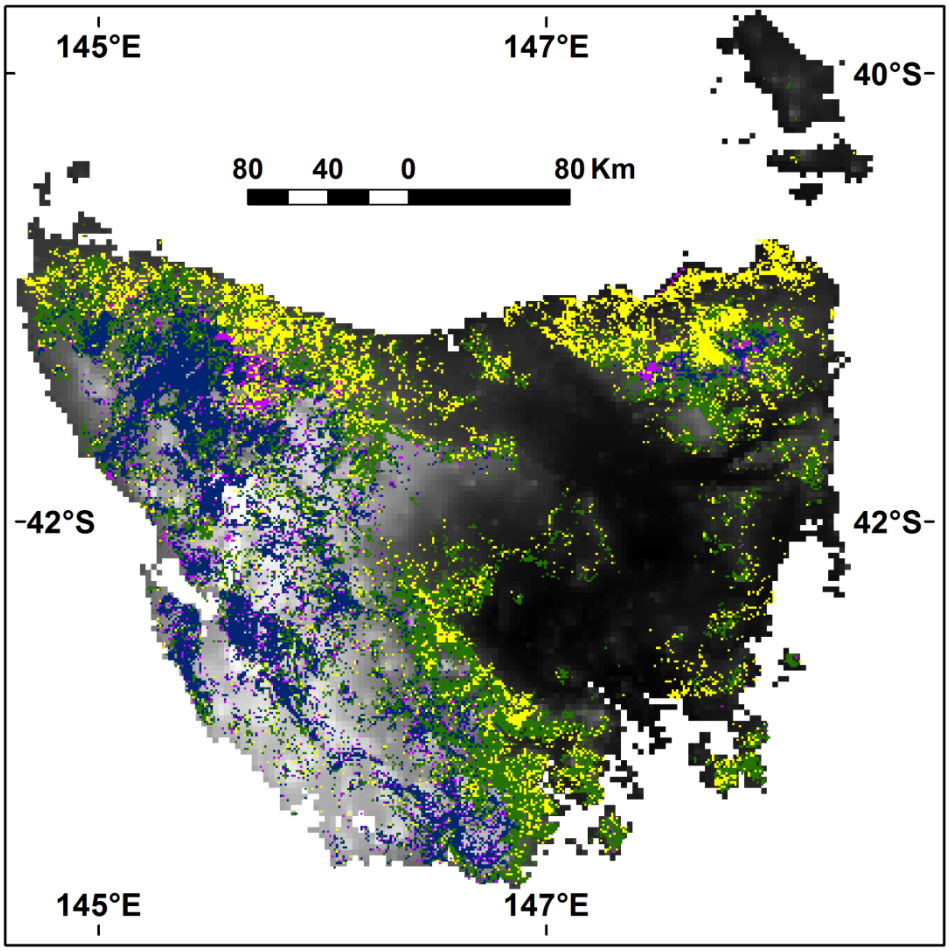


Figure 2-3. Distribution of eucalypt TOF and rainforest in Tasmania. Present remnant TOF— dark green, removed pre-European TOF— yellow, present remnant rainforest— dark blue, removed pre-European rainforest— purple. Forest data from NVIS (Commonwealth of Australia, 2012). Background, grey-level shading is mean annual rainfall, higher rainfall shaded lighter. Geographic projection.

Here an ecological basis for examining C stocks in primary TOFs is provided, with the SWA and Tasmania hotspots as exemplars. The biodiversity content of this chapter is in the form of review and synthesis. New scenario modelling of C forecasts is presented for two types of TOF in Tasmania under timber extraction, using CAR4D and FullCAM (section 1.3.6) with improved parameterisation for SOC. Parameterisation is re-checked in Chapter 4 where a sensitivity analysis is performed, and compared with other models in Chapter 7. The forecast temperature

increase for Australia, due to climate change, is 1–5º C over the next century (Williams et al., 2009). Forecasts are for humidity to decrease and rainfall to decrease by 10–20% in the southern forests (Mansergh and Cheal, 2007; Williams et al., 2009). The roles of these primary-forests, in terms of climate change mitigation and biodiversity refugia are considered in this study. New scenario modelling is also presented for the change in soil organic carbon (ΔSOC) in SWA and Tasmania under climate change.

2.3 Methods

Characterisation of TOFs in terms of C stocks is pertinent but only basic estimates exist at the regional level, such as for potential biomass within Australia as used in the FullCAM model (Waterworth and Richards, 2008) and for SOC within Australia (Barson et al., 2002). From such layers plus more elementary data— climate, climate change, forest cover and land tenure (from Bureau of Meteorology; CSIRO; Department of Environment, Water & Heritage; and Geosciences Australia respectively) statistics on carbon stocks and change were derived for the major C pools of the TOFs and rainforests using GIS.

The ΔSOC accompanying the temperature and rainfall effects of climate change was calculated but without CO_2 fertilisation. CO_2 fertilisation was not included as Australian native vegetation generally appears to have entered a phase of diminishment due to decreasing moisture and increasing temperatures, rather than extra growth due to increased CO_2 (e.g. Barber et al., 2000; Calder and Kirkpatrick, 2008; Allen et al., 2009). Comprehensive modelling of climate change found forest biomass to be most responsive to change in precipitation, and SOC more responsive to change in temperature (Cowling and Shin, 2006)— both would impact on the SOC levels for 2100. The future climate rasters were calculated using Ozclim (CSIRO, 2007) with settings from the climate change models 'ECHAM5/MPI-OM', 'Miroc-M', 'CSIRO Mk3.5' and 'ECHO-G'; with medium climate sensitivity and the 'A1B' and 'A1FI' emission scenarios. The choice of climate change models was for those

known to be most applicable to Australia (Perkins et al., 2007). The climate layers for years 2000 and 2100 were used in a temperature- and rainfall-dependent formula for SOC[‡] (Bird et al., 2002) to provide the relative change between 2000 and 2100. That relative change was applied to the SOC layer of 2000 (Barson et al., 2002) to give the SOC layer as initiated by climate change up to 2100.

2.3.1 Study areas

2.3.1.1 SWA

The SWAFR occupies 34.7 Mha (Fig. 2-1) and is species rich, with over 7,300 native vascular plant taxa (36% of Australia's vascular plant taxa), high levels of endemism (49% for vascular plants), over 2,500 species of conservation concern, and with significant phylogenetic diversity (Hopper and Gioia, 2004). There has been considerable speciation among some woody sclerophyll and herbaceous families from the mid-Tertiary period in response to progressive aridity. The southwest of the SWAFR contains all of Western Australia's ~190,000 ha of TOF. These forests average ~1,130 mm year^{-1} rainfall and experience a dry summer over 3–4 months. They are dominated by karri (*Eucalyptus diversicolor,* Figs. 2-4, 2-5), with small areas of tingle (*E. brevistylis*, *E. jacksonii*, and *E. guilfoylei*) in the south. The overstorey is characteristically uneven-aged (Fig. 2-4) as entire stands are rarely killed by fire (Wardell-Johnson, 2000). Intense fires lead to regeneration in gaps—perpetuating an uneven-aged stand, with higher intensity fires creating larger gaps. The dominant understorey species in these forests regenerate as a stand following fire (Wardell-Johnson and Williams, 2000).

[‡] SOC= -546.23- 0.0075609(MAT)^3- 134.54ln(MAP), adjR^2= 0.75, sigma= 74.34, SOC= soil organic carbon to 0.3 m below mineral soil surface in mg cm^{-3}, MAT= mean annual temperature in °C, MAP= mean annual precipitation in mm.

2.3.1.2 TAS

Tasmania covers ~7.2 Mha (including islands) of which 49% is currently forested, 3.1 Mha being native forest (MPIGA, 2008). The area of TOF in Tasmania has been reduced since ~1750 by ~30% (Fig. 2-3). Common eucalypt species in Tasmanian TOFs are *E. regnans*, *E. obliqua*, *E. delegatensis*, *E. globulus* and *E. viminalis*. In mixed-forest (Fig. 2-6) the rainforest understorey is categorised as 'M-'or 'M+' with heights of 8–25 m and >25 m respectively (Stone, 1998), and the eucalypts are generally >34 m. The mixed-forest and taller rainforest are generally found on moderate to high-fertility sites. In wet-sclerophyll TOF (Fig. 2-7) the eucalypts may be either even-aged or uneven-aged, with the rainforest understorey species often <12 m high, and the understorey is not successional with respect to the eucalypt canopy. Different forest types occupy an irregular spatial mosaic, evident at multiple scales, and acclimatised to the local environment and its fire history.

Figure 2-4. TOF of *Eucalyptus. diversicolor* (karri) and *E. jacksonii* (red tingle) showing effects of a high-intensity fire in 1951. Nornalup, SWA.

2.3.2 Tasmanian carbon accounting case studies

2.3.2.1 Scenario 1

The C trajectory was modelled using CAR4D (Dean et al., 2003; Dean et al., 2004; Dean and Roxburgh, 2006) for an even-aged, primary-forest, *E. regnans*-dominated mixed-forest, in coupe SX004C (described in Dean et al., 2003). (The eucalypt average height was 55–76 m, 'E1'; and the myrtle was 'M-'). It was converted by 'clearfell, burn and sow' (sowing of eucalypt seed) (CBS) (Ellis et al., 1982) to a production eucalypt forest with a harvesting cycle length of 80 years. (Planned rotation times for this forest type are 40 to 90 years, Kirkpatrick (1986)). The pools modelled were biomass (canopy and understorey), necromass, wood products (sawlog and pulpwood) and SOC. Conversion of primary forest to secondary forests entails a drop in SOC in the long term (several hundred years) of 6 to 48% but in the very short-term the flux direction can vary, depending on for example, silvicultural technique and degree of soil mixing (Pennington et al., 2001; Ferré et al., 2005). The effects are examined in detail in Chapter 7 and a prognosis is given. Attiwill and Adams (2008) suggested that 2% of the whole-profile SOC is lost during wildfire (without logging). The modelled ΔSOC here accompanying CBS in the very short-term, assumes a conservative 2.5% loss, distributed over the full soil profile (though most likely near the surface where the intense burn was administered). Any lateral transport of dissolved organic carbon (DOC) was modelled as remaining on-site as SOC.

2.3.2.2 Scenario 2

FullCAM (version 3.13.8, Waterworth and Richards, 2008) was used to model logging of wet-sclerophyll primary TOF in coupe HU320a. The forest comprised an uneven-aged and mixed-species canopy, dominated by *E. obliqua*, *E. ovata* and *E. amygdalina*, with relatively low, understorey biomass. It had been lightly, selectively logged in ~1950. Wood-product data were acquired for the clearfelling operation in

Figure 2-5. Even-aged regrowth karri forest maturing after clearfelling prior to 1900 in south-western Australia. Lower structural diversity than late-mature primary forest in Fig 2-4.

Figure 2-6. Mixed-forest TOF of *E. regnans* and well-developed rainforest understorey, Andromeda block, Styx Valley, Tasmania.

Figure 2-7. Wet-sclerophyll TOF, mixed-species and uneven-aged. Logging coupe HU320a prior to clearfelling for plantation, showing minimal understorey, typical of lower-biomass wet-sclerophyll. The Hunstman, Great Western Tiers, Tasmania.

1994 (Forestry Tasmania, personal communication, 1995) and conservatively expanded to give a pre-logging C in biomass of 116 Mg ha^{-1} (lower than average for wet-sclerophyll— Table 2-1). The primary forest was replaced by an *E. nitens* plantation (planting of germinated trees) with a typical 15 year (Greaves et al., 2003) plantation cycle. The default, non-endemic species multiplier of 2.3925 (Waterworth et al., 2007) was used for the plantation (it corresponds to a species-specific site-index). In FullCAM the forest vegetation is treated as a continuum with growth

based on local environmental conditions (soil and climate). FullCAM does not model ΔSOC due to mechanical disturbance during logging, nor emissions due to the intense regeneration burn (section 1.3.6). Consequently only the biomass, necromass and wood product pools were modelled. The ratio of sawlog to pulpwood volume from the primary-forest was 0.15. The wood product from the plantation was pulpwood.

2.4 Results

Carbon stocks both in the aboveground biomass and SOC pools follow a trend of increase from wet-sclerophyll forests in SWA, to Tasmanian wet-sclerophyll and mixed-forest, to Tasmanian rainforest (Table 2-1). This may in part be due to rainfall distribution and its relation to fire frequency. The SOC is approximately double or higher than that of the biomass. This latter trend however, may be a function of several factors such as actual environmental processes, the 'potential' (in FullCAM) being calibrated from a mixture of stand ages (including logged forest), or reliance on calibration from 2D remote sensing in FullCAM (i.e. discounts trunk length). Absolute values of potential biomass for wet-eucalypt in the NCAS have high uncertainty, and are underestimated (Richards and Brack, 2004; Keith et al., 2010), indicating they should sometimes be increased. However, such scaling information is unavailable for these broader regions of SWA and Tasmania. Owing to the lack of extensive, local calibration in deriving the base layers used, the results give an indication of the order of magnitude, and error margins may be ~50%, with the precision for SOC being lower than for biomass. Additional C, ~13% of the biomass, is likely to be in the CWD pool.

The SOC loss with climate change (Table 2-2) is consistent and high (~30% initiated by climate change to 2100) for SWA, which concords with the drying trend since 1970 in SWA (and hence lower NPP). The SOC efflux for Tasmania rainforest is also reasonably consistent, but less than that for SWA. The ΔSOC initiated by climate change to 2100 is much less consistent between models for TOFs in

Tasmania. This arises from different climate projections having higher or lower rainfall on different sides of TAS; and because the TOF zone runs mainly down the middle of the island: the average effect is close to zero change. Using the CSIRO Mk3.5 model the SOC emission from TOFs in Tasmania is 5–10%.

Note that this modelling of temporal change, using a formula designed for spatial variation, constitutes substituting time-for-space. This approximation may be true to some degree but is a simplification of the ecosystem dynamics of climate change, and therefore these figures are only a guide to possible trends, rather than actual quantitative forecasts. Also, different SOC pools change their C efflux at different rates with climate change (Knorr et al., 2005) and therefore these SOC fluxes may be more of a long-term, time-averaged asymptote of a virtual, stable climate, rather than a real endpoint achieved over the next century.

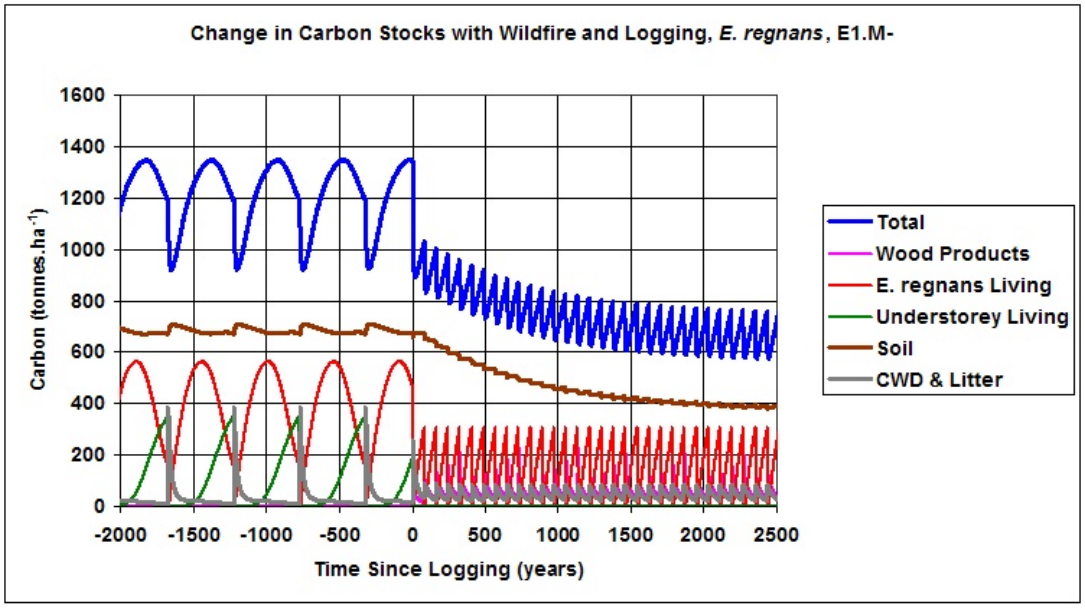


Figure 2-8. Scenario 1. Carbon stocks (including wood-products) for TOF primary-forest conversion. *E. regnans*– dominated primary TOF (55–76 m high) with low myrtle understorey biomass, converted to an 80 year *E. regnans* timber-harvesting cycle. Software used was CAR4D.

Within about 25 years of logging primary TOF in scenario 1 the C efflux is 450 Mg ha^{-1} (average of 26 Mg ha^{-1} $year^{-1}$, Fig. 2-8). There are further net emissions with successive harvests until the system approaches an asymptote towards 2,000 years. Long-term C stocks (in the absence of climate change effects) are 1,210 Mg ha^{-1} and

658 Mg ha^{-1} for the mixed-forest and production forest (including wood products) respectively: a difference of 552 Mg ha^{-1}.

One year after initial logging in scenario 2 the C efflux is 35 Mg ha^{-1} and by 2016 it is 64 Mg ha^{-1}, followed by sequestration as the *E. nitens* stand grows. The successive years of wood products do not prevent a long-term net emission compared with the C stock of the primary forest (Figure 2-9). The long-term difference in standing stocks of C (excluding SOC, which was not modelled in scenario 2) is 83 Mg ha^{-1}. In Australian forests SOC is ~50% of the total C stock (MPIGA, 2008, p117). If the long-term ΔSOC is roughly taken pro-rata with change in biomass then the long-term change in total C stock for scenario 2 is ~167 Mg ha^{-1}. Note that the conservative estimate of pre-logging biomass, in combination with the use of the default non-endemic species multiplier for the plantation, portrays the emissions for this primary forest conversion to plantation conservatively (as knowledge of the site indicates that it is likely to be poorer than the multiplier portrays).

The typical situation for Tasmanian logging of primary TOFs is likely to lie somewhere between scenario 1 and scenario 2 (due to their diverse locations, site-histories and silviculture types), i.e. long-term C emissions of between 167 and 550 Mg ha^{-1} (averaging 358 Mg ha^{-1}). Note however, that scenario 1 is not at the upper limit of emissions as the rainforest was not M+ type as in Dean et al. (2003); also, in scenario 2 the original forest was below average for Tasmanian wet-sclerophyll biomass, even according to FullCAM (Table 2-1). Therefore both these emission scenarios are conservative.

The last forty years in Tasmania have seen similar area-totals logged annually (e.g.: ~2,500 ha of primary-forest logged in 2004-2005, Forest Practices Authority Tasmania, personal communication, 2011) and therefore similar emissions. If similar amounts are thus logged in the ensuing 20 years, to achieve the stated 35% of sawlog quota from primary forests, then it follows that its long-term addition of C to the atmosphere will be in the order of 17.9(±9.0) Mg (i.e. ~66(±33) Mg of CO_2

Table 2-1. Derived statistics for SWA and Tasmania. ('Formal Reserved area' includes only reserve types: WHA— World Heritage Area or NP— national park. Derived values were calculated from data layers in GIS.)

Forest type	Pre-European area (ha)	Current Area (ha)	State Forest	Freehold	Formal Reserved area, %	Mean Rainfall (mm $year^{-1}$)	SOC to 0.3 m (Mg ha^{-1})	C in potential aboveground biomass (FullCAM) (Mg ha^{-1})
TOF SWA	208,000	190,161	60%	20%	20% , NP	1,132	123	71
TOF TAS	1,360,000	814,000	54%	17%	14%, WHA	1,559	271	92
Rainforest TAS	785,000	593,000	30%	4%	34%, WHA	2,218	369	100

Table 2-2. Forecasts for SOC to 0.3 m. Change in SOC induced by change in biomass to 2100, initiated by climate change.

Forest Type	CSIRO Mk3.5 (Mg ha^{-1})		ECHAM5 (Mg ha^{-1})		ECHO-G (Mg ha^{-1})		MIROC-m (Mg ha^{-1})		Average model (Mg ha^{-1})		Average model, C area totals (Mg)		Average model change (%)	
	A1B	A1F	A1B	A1F	A1B	A1F	A1B	A1F	A1B	A1F	A1B	A1F	A1B	A1F
TOF SWA	94.942	89.233	86.350	75.406	85.097	73.413	92.680	86.145	89.767	81.049	-6.35	-8.01	-27	-34
TOF TAS	256.96	241.70	276.44	272.12	271.35	264.32	283.71	282.60	272.12	265.19	0.93	-4.71	0.4	-2.1
Rainforest TAS	317.47	301.22	335.83	330.09	330.80	322.30	345.23	343.53	332.33	324.28	-21.51	-26.29	-10	-12

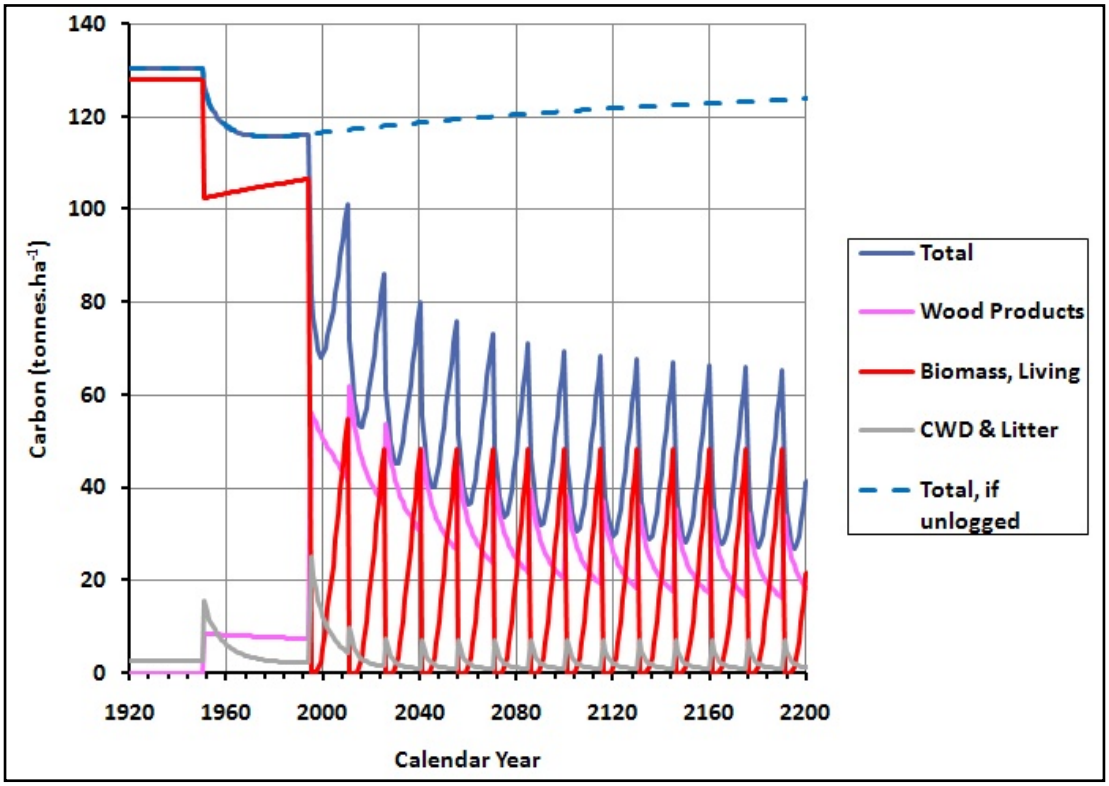


Figure 2-9. Scenario 2. Carbon stocks (including wood-products) for conversion of wet-sclerophyll, primary TOF to plantation. Logging coupe HU320a, clearfelled in 1994, northern Tasmania. Plantation species *E. nitens*. The forest had been lightly selectively logged in ~1950 (for sawlogs) and was still recovering in 1994. Software used was FullCAM (ΔSOC was not modelled).

equivalents). The long-term C trajectory under logging will of course be affected by climate change.

2.5 Discussion

2.5.1 LUC effects on carbon stocks

There are three aspects of primary forests pertinent to C and climate change: (a) the present contribution towards keeping C out of the atmosphere, (b) the impact of their removal upon atmospheric C concentrations, and (c) the impact of climate change on their role as a C store. Although focus is on C because CO_2 is the main contributing

GHG, the importance of other GHGs (e.g. NO_x and H_2O) and their constituent elements in forest biochemistry is also recognised.

Oldgrowth and late-mature primary forests can absorb atmospheric C and thereby contribute to climate change mitigation (Phillips et al., 2002). Although labelled oldgrowth or late-mature-primary-forest (as they are considered undisturbed), their net C influx may be due to ongoing recovery from past anthropogenic disturbance (Heath et al., 1993). This concurs with the finding of higher carbon stocks in late-mature-primary-forest than in mature forests (Keeton et al., 2010). However, caution is necessary when attributing sink status as some measurements of net sequestration in primary-forests have used flux towers, the singular use of which can neglect the fate of DOC and volatile organic compounds, which could alter the balance between sequestration and emission. Many C forecasting models assume a C flux equilibrium (or 'carbon-cycle steady state') on time-scales of a century to a few millennia, but equilibrium is only one of an infinite number of flux magnitudes. Assumptions of equilibrium can lead to the calculation of dubious SOC stocks and growth parameters (Carvalhais et al., 2008). Examples of non-equilibrium, in the form of net sequestration are: 1) fossil fuel deposits; 2) illuvial and benthic layers of organic residues; 3) the tundra with its climatically-reduced organic decomposition in subsoils; and 4) peat layers derived from forest biomass, for example the deep layer burnt after logging the primary-forest spruce of Dolly Sods, West Virginia (Clarkson, 1964). Determining the net C flux of primary-forest forests would help to identify significant vectors of the C cycle and encourage the development of appropriate policy and management in relation to climate change.

The low precision in C flux modelling of primary-forest logging results from variation and uncertainty in: 1) historical fire frequency; 2) age at which the primary-forest is felled; 3) environmental site quality; 4) age of onset and rate of senescence; 5) understorey biomass and successional capacity; 6) severity of soil disturbance (e.g. differentiating between efflux from SOC, and influx from debris and ash); 7) decay rates of logging and milling waste; 8) effects of future climate change on growth and decomposition; and 9) uncertainty in current stocks owing to lack of

allometrics for mature and non-target plants. (Beaumont et al., 2007) modelled the effect of climate change on biodiversity to determine uncertainty levels. They found large differences in forecasts between models, and between starting conditions—requiring forecasts from multiple scenarios and models. This is one reason why different models and scenarios were chosen for C forecasts.

GHG emissions due to timber harvesting (with regeneration burns) are likely to be higher than for natural disturbance owing to the extra mechanical disturbance, the more intense fire, and the short half-life of mill residues. Accounting for wood products however, does not indicate that timber harvesting is better for net C sequestration compared with leaving the forest unmanaged (Seidla et al., 2007). Instead, the results indicate significant emissions from logging primary-forest in Tasmania and that the present management of primary-forest C is pertinent to at least the next two millennia. History has shown that anthropogenic C emissions build over millennia, e.g. to generate our current climate change predicament. In Chapter 4 a sensitivity analysis is performed to show the variation in wood-product half-lives necessary to maintain the C store of the primary forest. The long-term loss accompanying the LUC to harvesting cycles is principally because the half-life of biomass is longer than that of necromass, necromass decays more slowly as forest debris than as mill residue, soil disturbance is less in wildfire, and major fires are less frequent in the primary forest.

Logging in SE Australia leaves 30–50% of the above-ground biomass as *in situ* 'forest residue' ('slash') (Fig. 2-10; Ximenes et al., 2008a) to be burnt, with further wastes in downstream processing. The amount of 'forest residue' is age dependant, being higher for primary-forest and for immature individuals (Ximenes et al., 2008a), and includes the non-target species. Also the ratio of sawlog to pulpwood retrieved can be as low as 0.04 (unpublished data from private forests) and pulpwood has a much shorter half-life, thereby contributing to climate change. Thus, current forest management and resource usage impose a short half-life on the solid phase of C in primary-forests.

The silvicultural prescriptions applied to public forests in Tasmania may more frequently include alternatives to clearfell such as aggregated retention (Forestry Tasmania, 2005; Forestry Tasmania, 2008), for which the burn intensity is lower (Scott et al., 2012) and thus C efflux (per hectare) is lower. However biomass of the new eucalypt stock may also be lower (Neyland et al., 2009); consequently its deployment is uncertain. Nevertheless similar emissions will still arise from several of the pools when considering the broader landscape (i.e. Tasmania as a whole), as the acreage of forest felled must be similar, because the forecast annual product quota has not been reduced. Also, there was no indication of reduced application of CBS on private land.

Figure 2-10. Logging of Mixed-forest, type M- with *E. obliqua* and *E. regnans* ~45 m high. Logging coupe WE008e near Mt Wedge, Tasmania. The foreground shows the typical substantial amount of 'forest residue' (slash) produced, which forms a large portion, 30–50%, of CWD (Ximenes et al., 2008a) burnt during CBS and results in high carbon efflux. For scale, note large vehicles on road, including cable-logging tower in background.

2.5.2 LUC effects on biodiversity

Within the landscape, it is important to distinguish regenerating forests after logging, from areas of natural regeneration (young primary-forest) reserved from timber harvesting. The former are not scheduled to become oldgrowth, instead being felled while young, in the ongoing harvesting cycle (Forestry Tasmania, personal communication, 1996). An unmanaged landscape will comprise: even-aged stands of various age cohorts, uneven-aged stands, and spatially-diverse successional processes. Demographics will be in some form of balance at the broader landscape level. The relative proportion of even-aged to uneven-aged stands is a function of canopy species, location and land-use. For example in TAS: *E. regnans* stands are more likely to be even-aged than are *E. obliqua*; *E. regnans* stands are more likely to be uneven-aged than those on mainland Australia (Turner et al., 2009); and a majority of wet-sclerophyll forests in SE Australia are uneven-aged. However, this latter trend only represents statistics for the vestigial primary-forest, with perhaps a higher proportion of even-aged primary-forest removed since 1750, owing to the preference of primary industries for the richer, less fire-affected land. For karri TOFs in SWA a considerable portion of uneven-aged forest (about 60,000 out of 190,000 ha) has been converted to young, even-aged stands, particularly during commercial clearfelling from 1966–2000, with the uneven-aged primary-forest now mostly in road and riparian reserves, and national parks etc.

Primary-forest functions include the provision of a diversity of structural forms containing a variety of micro-habitats not available in the structurally-simpler, artificially regenerated stands (e.g. Burrascano et al., 2008; Liira and Kohv, 2010; Manes et al., 2010). There is greater variety of foraging and nesting substrate, and continuity of niches for wildlife in primary-forest, although some species with wide home ranges may also rely on the variety within the wider environment (e.g. Wardell-Johnson & Williams 2000, Bekessy et al 2009; Fig. 2-4). For example, avian foraging specialists such as the Crested Shrike-tit (*Falcunculus frontatus*) and Rufus Tree-creeper (*Climacteris rufa*) are rare in even-aged karri stands, but are associated with primary-forest. Nevertheless, most vertebrate species which occur in

primary-forest karri also occur in even-aged karri regeneration (Fig. 2-5). In addition Gondwanan relictual invertebrates (e.g. the spider taxa *Chasmocephalon* sp., *Dardanus* sp., *Baalebulb* sp. and *Moggridgea tingle*) and small vertebrate taxa (e.g. the frog *Metacrinia nicholsii*) persist in moist, protected habitat of late-mature primary-forest (Wardell-Johnson and Horwitz, 1996). They are vulnerable to the major disturbance of clearfelling and regeneration, but persist following high-intensity fires in primary-forest. Nevertheless, many elements of the biota restricted to refugial habitat in TOF and rainforest became extinct in SWA during the Pleistocene (e.g. the rainforest tree genera *Dacrydium*, *Phyllocladus*, *Nothofagus* and *Athrotaxis*) and changed fire regimes are implicated (e.g. Hopper, 2003). The continued depletion of primary-forest will incur attrition of native species (Bekessy et al., 2009; Bradshaw and Ehrlich, 2015).

A shift in biomass and structure, but not in overall floristic species richness or community type has already been observed with the introduction and intensification of *Phytophthora* (causing dieback, especially in jarrah (*E. marginata*) in SWA, Shearer et al. 2004). Spread and intensification was largely through road building, especially after WWII (Calver and Wardell-Johnson, 2004). Consequently, jarrah open-forest was replaced with jarrah-marri (*Corymbia calophylla*) woodland, or on lower slopes, bullich (*E. megacarpa*), and the ground layer of vegetation became more open; all these changes also reducing C stock. Most TOF is not considered susceptible to *Phytophthora*. However, a reduction in microflora or a more open understorey associated with overstorey changes could shift that balance.

2.5.3 Effect of climate change on carbon stocks

Trends for climate change impacts in Australia have been reviewed (e.g. Mansergh and Cheal, 2007; Williams et al., 2009) and general conclusions are: 1) increased temperature may increase decomposition rates and hence C efflux; 2) more frequent fires; 3) changing vegetation will affect SOC; 4) ultimate control over SOC stocks may be through fire and land-use change management; 5) some trees will be stressed

due to temperature increases and sporadic water deficit whereas others may benefit from reduced frosts; 6) disease, insect attacks, weeds, higher storm intensity, increasingly episodic rainfall and increased fire frequency may deplete stressed forests, lowering their productivity; 7) there will be increased pressure on survival of some species and ecological communities; and 8) more CO_2 may accelerate plant growth or decrease growth by disrupting ecophysiology but the change in temperature and rainfall patterns is expected to dominate (hence the focus when modelling ΔSOC).

The analysis of ΔSOC with climate change presented here generally indicated efflux, with the highest percentage change being for SWA (which follows the already-evident drop in rainfall), followed by that for the Tasmania rainforest, and the lowest change being for Tasmanian TOF. However, there was high variability between climate models for the latter forests. Climate-change-induced drought has already caused decline of trees in central Tasmania (Calder and Kirkpatrick, 2008), indicating a net C efflux. The highest magnitude of emission from SOC will be from rainforests, this being similar to the long-term emissions from primary-forest logging planned for public forests in Tasmania.

For karri TOFs the higher fire intensities and frequencies with climate change, suggests a shift from karri to karri-marri and from karri-marri to jarrah-marri forest, i.e. decreasing biomass. Higher fire frequency and intensity in Tasmania shifts ecosystem types from rainforest to mixed-forest to wet-sclerophyll to dry-sclerophyll (Gilbert, 1959; Cremer, 1960), which incurs an average decrease in C stocks (Table 2-1). Other types of ecosystem change can occur only with more severe change in fire regime, e.g. woodland-to-open savannah, and increased grass/tree ratio (e.g. Cloudsley-Thompson, 1975), thereby inducing higher C emissions. Where more-frequent fire changes ecosystems to those with vegetation that survives fire, then the new vegetation is usually more flammable and conducive to fire propagation (Kirkpatrick, 1986); or at least its aboveground component is.

Carbon emissions from non-stand-replacing fires of moderate intensity can be in the order of 34% of biomass C, and post-fire sequestration is limited more by tree mortality than by survivor productivity (Irvine et al., 2007), and significant emissions come from decaying roots after tree mortality. Emissions from more-intense fires are up to three times as high as those from moderate fires (Irvine et al., 2007).

Frequent fires can reduce site-quality, and consequently long-term C storage, and may increase erosion, providing positive feedback to climate change (e.g. Gough et al., 2007a; Williams et al., 2009). Soil nutrient losses are higher with more intense or higher frequency fire, and feedback leads to a more fire-prone ecosystem with reduced soil nutrients (McIntosh et al., 2005). Over several millennia the process can dramatically alter the soil type, for example by reducing clay content by eluviation (McIntosh et al., 2005). Examples of other positive feedbacks to climate change are: reduced forest biomass and reduction in locally-induced rainfall (Li et al., 2000); a higher proportion of more labile SOC (Gill, 2007); and increased drought severity or frequency with climate change, causing emissions from biomass (Warszawski et al., 2013).

The forecast reduction in SOC is thus due to drought stress and higher fire frequency affecting vegetation type, and thus in turn, the soil. Earlier increased growth due to higher atmospheric CO_2 concentration (e.g. Berry and Roderick, 2006) (corresponding to net sequestration) would be counteracted by the drop in SOC. Climate modelling, more detailed than the present work, has confirmed the forecast an increased fire danger index forecast for Tasmania this century, especially for the southern region which contains the most carbon-dense TOFs and largest trees (Fox-Hughes et al., 2014). Prescribed burning has been proposed to reduce readily combustible material (so-called 'fuel'[3]) in forests (and thereby increase climate change resilience) but modelling and evidence suggests that it would not reduce

[3] The terminology which comes from firefighting refers mainly to dead, dry aboveground biomass but can be subjective, and has been adopted in applied science and industry. From a physical science perspective anything combustible under appropriate conditions is fuel, e.g. all living and dead biomass.

carbon emissions (Bradstock et al., 2012) and that 'fuel' and fire management close to human infrastructure would be more fruitful (Enright and Fontaine, 2013). Limiting forest fragmentation and general anthropogenic disturbance is likely to help maintain existing mesic micro-environments that protect 'fuel' against drying-out and thus fire, in mixed-forests. The importance of primary forests in climate change policy (including those of TOF) has been re-iterated, with policy initiatives broached for their conservation (Mackey et al., 2015).

In line with the longer half-life of SOC than biomass C it must be made clear that, although the declining SOC stock was modelled by correlation with climate changes occurring by 2100, SOC was only a proxy for biomass C (section 7.5), with the decline being based on response to a changing climate, and that change in biomass C will occur long before corresponding ΔSOC, at a timescale an order of magnitude different (100 years compared with 1,000 years). The direct effect of warmer temperature on SOC itself is a different matter however, with possibly a more spontaneous reaction.

Exotic plant species can out-compete native species, accrue high fuel loads, change fire frequency, increase the area burnt, increase C emissions and reduce aboveground biomass (e.g. Litton et al., 2006). Venevsky et al. (2002) showed a logarithmic increase in fire frequency with human population density. The more sparsely-populated regions of Australian are presently at the most responsive part of that curve. Under climate change, the effects of exotic plants and fire-induced change, may synergistically impact on biodiversity and C stock.

In summary, climate change is likely to cause net emission by converting forest types and through positive feedback. Human population increase will also inflict attrition on forests. The function of primary-forest forests under climate change will thus be influenced by people's impacts upon them today.

2.5.4 Management opportunities

Uneven-aged primary-forest generally has higher biomass and total-system C than even-aged primary-forest, notably when the older individuals are 'silviculturally overmature' (a forestry term) (Kurz et al., 1996; Phillips et al., 2002). The rainforest species in mixed-forests could be a biomass-counterpart to the younger eucalypts of uneven-aged, wet-sclerophyll forests. Thus the mixed-forests are worthy of further study to gain more precision for their C dynamics, and careful management, at least for climate change mitigation and ecosystem conservation.

Prescribed burning can influence carbon stocks. Prescribed burning of TOFs will be increasingly ineffective at protecting infrastructure during severe fire weather, as such events are forecast to increase with climate change, regardless of necromass levels (Williams et al., 2009). Instead, targeting areas of lower C stocks, greater flammability and ecological resilience, in neighbouring vegetation (such as planted grain crops and pastures), can help maintain tree biomass (Edwards et al. 2008).

The capacity of the world's vegetation to absorb emissions is currently exhausted (CSIRO, 2009). The impact of further anthropogenic emissions from attrition of primary-forest will therefore incur a more noticeable impact in the future, which increases the importance of avoiding such activities. In the longer-term, emissions from SOC associated with conversion of primary forests to short-rotation timber crops will have a substantial impact on the global C budget. Of course this finding does not detract from the [beneficial] climate change mitigation offered by plantation establishment on old-field farmland. The C in many primary-forest forests currently lacks economic value, as it is not recognised in legally-binding intergovernmental schemes, thus supporting replacement of primary-forest by plantations or short-rotation native forests. However, voluntary C markets, although with lower C prices, can help retain the C-reservoir function of primary-forest by offering financial incentives for its conservation.

Chapter 3 Pre-logging carbon accounts in late-mature primary-forest, via allometry: an example of mixed-forests in Tasmania

This chapter is adapted from a published paper: Dean et al. (2012a)

3.1 Abstract

Uncertainty in past and future anthropogenic carbon emissions obscures climate change modelling. Available allometrics are insufficient for regional-level accounting of the pre-logging carbon stocks of most late-mature primary forests. The project goal was to determine the aboveground carbon (biomass and necromass) for a typical oldgrowth *E. delegatensis*-dominated mixed-forest in Tasmania. Allometrics were developed for aboveground biomass of *E. delegatensis* and generic rainforest understorey species. A total of 207 eucalypts with DBH 0.21–4.5 m, and 897 rainforest understorey trees with DBH 0.01–2.52 m were measured across 7.7 ha. DBH frequency distribution of *E. delegatensis* showed at least two age cohorts and distinct positive skew, whereas its DBH carbon distribution showed distinct negative skew. Half of the eucalypt biomass was from trees with DBH>2.4(0.1) m, and 16% with DBH≥3.5 m (from ~1.1 trees ha^{-1}), indicating the importance of allometrics for high DBH. Aboveground carbon was 636(172) Mg ha^{-1}, with ~23% from understorey and ~24% from necromass. The carbon in aboveground biomass was above the median value for temperate forests. The long-term aboveground-carbon emissions from LUC by clearfelling the same forest type from 1999–2009 is likely to be 2.9(±1.3) Tg, depending on the growth and seral stages of the forest logged.

3.2 Introduction

Accounting of forestry's industrial carbon emissions is important in determining the carbon fluxes accompanying present and future industrial activity and in climate change modelling (Chapter 1). The interacting effects of fire and logging that influence the landscape mosaic of mixed-forests (Lindenmayer et al., 2009), have spatio-temporal implications for determining carbon dynamics. Determining stand-level carbon stocks in forests of varying seral stages is a crucial first step to investigating landscape-level, long-term carbon dynamics.

In the previous chapter it was shown that modelling of clearfell logging in primary wet-eucalypt forests indicated a long-term loss of approximately half the original carbon stock, when the LUC was to either plantation or to native eucalypt secondary (both on harvesting cycles). The primary wet-eucalypt forests, have long-been a prime source of pulpwood and high quality sawlogs (Felmingham et al., 2004; Elliot et al., 2008). Conversion of these primary-forests has been at the industrial scale since the 1940s (Helms, 1945) and clearfelling of the mature remnants on public land allocated to forestry ('State' forests) is due to be phased out by 2030 (Forestry Tasmania, 2009c). From the 1960s, production increased and pulpwood export dominated (Florence, 1993), multiplying at least fourfold from 1970–1980 to ~2.2 Tg $year^{-1}$ (dry weight) and remaining near that level to 2000 (Tasmanian Government, 2006).

The magnitude of the long-term (~2,000 years) emissions depends in part on the original long-term means of the C in the pre-logged forest, the secondary forest and in wood products. Developments in forest inventory in Australia have concentrated mostly on post-harvest stands up to merchantable age. Consequently, allometrics for trees older than about 100 years are rare. Allometrics suitable for carbon accounting were developed for all growth stages of wet-*Eucalyptus regnans* F. Muell. (swamp gum or mountain ash) forests with eucalypt DBH up to 6.44 m (Dean et al., 2003; Dean and Roxburgh, 2006) thus enabling both long-term and short-term C flux forecasts. Primary-forest dominated by *E. regnans* has been a prime target of

industry in the Florentine and Styx valleys (Tasmania), although the closely related wet-*E. delegatensis* R. Baker (white-top/gum-top stringy bark, alpine ash) and wet-*E. obliqua* L'Hérit. (brown-top stringy bark, messmate) mixed-forests were also logged (Australian Newsprint Mills 1960s coupe inventories, unpublished, Forestry Tasmania; Walker & Felton 2007). In Tasmania *E. delegatensis* was previously known as *E. gigantea* f. Hook (ANM, c1960). Logging of primary wet-eucalypt forest for pulpwood and sawlogs is ongoing in State forests and on private land across Tasmania, although the pulp market has been superseded to some degree by eucalypts grown in tropical climates. Nevertheless, industrial-scale logging for bioenergy and sawlogs, sourced from secondary forests, is promoted (Dakis, 2013; Rothe et al., 2015). Depending on the political atmosphere, primary forests may also be logged (Fromberg, 2015). Therefore the present analysis is relevant to areas of both past and future LUC of primary forests.

The aim of the chapter is to provide the first account of aboveground (biomass and necromass) carbon for a typical primary *E. delegatensis* mixed-forest stand, prior to logging, based on field inventory. This entails construction of applicable allometrics, developed from state-of-the-art procedures and the currently, publically available literature and software. As error margins are important in interpreting carbon accounts, the errors in aboveground carbon stock are also estimated. Representativeness of likely emissions is also presented, from a review of logging records. Although SOC can substantially decrease after several logging cycles (e.g. Ferré et al., 2005; Tang et al., 2009; Vario et al., 2014) this decline is not examined in this chapter.

3.3 Methods

3.3.1 Study site

The study site was located in the Florentine valley of southern Tasmania. The geology, climate and forest ecology of the Florentine valley are described in Gilbert

(1959). The valley is in the southern-central distribution of wet-*E. delegatensis* forests in Tasmania, with a high-rainfall region of rainforests to the west, and agricultural areas and drier forests to the east (Figure 3-1). Although the wet-eucalypt forests in the mid-elevation Florentine valley are dominated by *E. regnans* (Gilbert, 1959) the upper Florentine wet-eucalypt forests are dominated by *E. delegatensis*, *E. obliqua* and *E. regnans* (approximately 1,500, 800 and 350 ha respectively). The study site was centred in the upper Florentine Valley at 42 44′ 36.10” S, 146 24′ 41.06” E, within the planned logging area (coupe) FO044A, of ~60 ha. Mean annual precipitation at the study site was 1600 mm year^{-1}, and elevation across the site was 448 m in the north to 505 m in the south. Ground slope within data collection plots, averaged 5(2)º. The study site was on Ordovician limestone, overlain in parts with Holocene sediments.

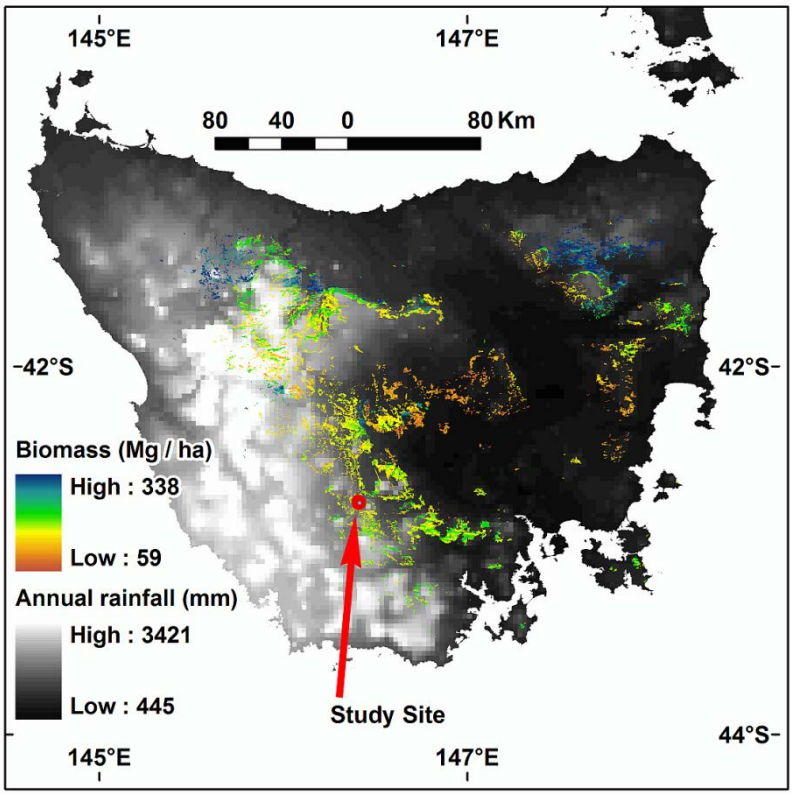


Figure 3-1. Distribution of *E. delegatensis* wet-sclerophyll mixed-forest from TASVEG, overlaid on mean annual precipitation. The colour scheme for *E. delegatensis* was from the FullCAM layer of potential aboveground biomass (Richards and Brack, 2004). Geographic projection.

At the landscape-scale the mixed-forest was interspersed with rainforest and the more-frequently burnt, wet-sclerophyll forest (Figure 3-2). The northern part of the coupe (wet-sclerophyll forest) had been subject to moderate or low-intensity wildfire in the last hundred years. The forest in the remainder of the coupe (with rainforest understorey trees present) appeared to be post-wildfire regeneration following a fire several hundred years ago and had not been subject to fire since. The site had not been logged prior to 2009.

The density of eucalypt trees in the primary *E. delegatensis* mixed-forest at the study site varied from under 5% to over 40% crown cover (Figure 3-3). Mean dominant height of the mature eucalypts ranged from approximately 41–76 m (from Forestry Tasmania's photo interpretation). The rainforest understorey was dominated by *Nothofagus cunninghamii* (Hook. f.) Øerst. (myrtle), *Atherosperma moschatum* Labill. (sassafras) and in some locations *Phyllocladus aspleniifolius* Labill. (celery-top pine) (Figure 3-4). There was a low abundance and cover of other typical understorey species: (*Acacia dealbata* Link. (silver wattle), *Nematoleposis squamea* Labill. ssp. *squamea* (lancewood or satinwood) and *Acacia melanoxylon* R. Br. (blackwood)). Myrtle trees reached up to ~40 m. This composition is typical of mixed-forests in Tasmania. This forest type can be described as a fire-maintained disclimax (Barker and Kirkpatrick, 1994).

3.3.2 Field data collection and processing

The centres of ten circular plots for data collection were positioned randomly within the unlogged portion of the operational boundary of the coupe. Each plot comprised three nested sub-plots of 7, 30 and 60 m radius. The diameter at breast height (1.3 m) (DBH) was the primary data type recorded. Other data recorded were species, growth stage (regrowth, mature or senescent), trunk hollow size, burl size, evidence of fire or adventitious roots. Trees with DBH ranges from 0.01–<0.2 m, 0.2–<1.0 m and

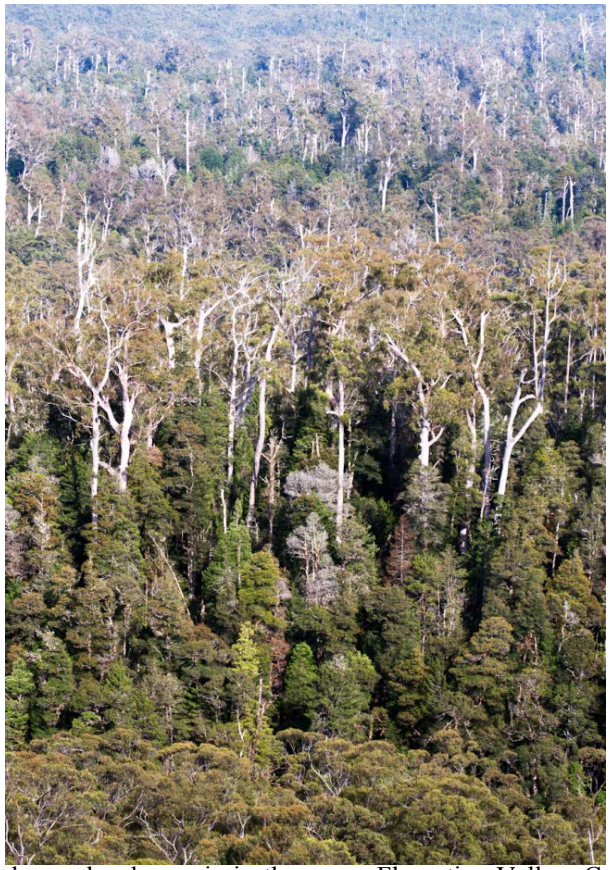


Figure 3-2. Landscape-level mosaic in the upper Florentine Valley. Cool temperate rainforest in riparian zones, juxtaposed with mixed-forest— where rainforest persists beneath emergent *E. delegatensis*. Photo: N. Fitzgerald, 2010.

Figure 3-3. Aerial photograph, upper Florentine Valley in the foreground. Looking approximately north. Shows both senescent and intact *E. delegatensis* over rainforest understorey in primary forest. Pale-green, conical-shaped trees are sassafras. Photo corrected for perspective distortion. Original photo: R. Blakers, 2010.

Figure 3-4. Primary mixed-forest in the southern part of the study area. Viewed from within recent clearing at the edge of the aggregate retention operation; showing spatially dispersed, *E. delegatensis* (both senescent and intact) trees over a dense rainforest understorey of varied-sized myrtle (darker, mid-height trees) and sassafras (lighter, shorter trees). Photo: N. Fitzgerald, 2010.

>1.0 m were recorded in the 7, 30 and 60 m sub-plots respectively. Dimensions of necromass (snags, and fallen logs and branches (coarse woody debris, CWD) with at least one diameter >0.2 m were recorded in the 30 m plots. Seven DBH-tree-height pairs were measured using a clinometer and laser rangefinder, for comparison with reported sizes elsewhere.

Data were collected by teams of numerous volunteers (i.e. 'citizen science' as described in Chapter 1) from mid-2009–2010, and recent logging had terminated when fieldwork began.

Due to resource constraints four of the plots could only be measured out to 30 m, and in one plot the 60 m circle partially overlaid a clearfelled area. Spatial density data were corrected for these variations during processing. Plot area was also corrected for the average ground slope across each plot. The total area sampled (after corrections) was 7.656 ha.

Allometric equations were used to estimate the biomass for each tree based on the measured DBH. This value was then discounted to account for decay in mature and senescent trees. Biomass of dead wood was estimated by calculating the volume of the log and discounting for decay.

For necromass, heights, lengths and diameters were recorded. The size of hollows were also measured, or estimated, where out of reach. A decay class was recorded, being one of H, M and S (hard, medium and soft) corresponding to 100%, 66.66% and 33.33% dry mass present for the corresponding volume. That decay class was also used in the field to represent voids, that were either too numerous or irregular to be measured as hollows.

3.3.3 Allometrics

The two main approaches for deriving tree biomass (or carbon) allometrics are destructive sampling (including weighing and carbon assay) and taper formulas. Information was combined from published reports that had used both of these methods. A commonly used approximation where carbon assay is not performed, such as part of destructive sampling, is to assume that C is 50 wt% of dry matter (biomass or necromass). The measured fraction can vary from 42% to 61%, depending on species, plant component and environment (Thomas and Martin, 2012) but 50% is a commonly used proportion (Gifford, 2001), and is a compromise

between the IPCC's 47% (Aalde et al., 2006) and the effect of the likely contribution from volatile compounds lost during typical assay (Thomas and Martin, 2012). For wet-sclerophyll *E. obliqua* in Tasmania, Ximenes et al. (2008a) found the fraction in stem cross sections to be 0.497(0.004), being close to 0.5— which was used throughout in the present work.

3.3.3.1 *Eucalyptus delegatensis* allometric

Allometrics for mainland populations of *E. delegatensis* ssp. *delegatensis* are not differentiated here from those for the Tasmanian endemic subspecies *E. delegatensis* ssp. *tasmaniensis* Boland, due to similarity in growth habit.

The available biomass allometric for the dominant canopy species (*E. delegatensis*) covered the DBH range 0.119–0.832 m (Raison et al. unpublished, in Keith et al., 2000), which was too limited, as the mature trees in the Florentine reached over 4.5 m. Merchantable volume allometrics covered the DBH range 0.616–1.656 m (Wang and Hamilton, 2003). However, biomass allometrics for the related ash species, *E. obliqua* and *E. regnans*, covered the DBH ranges 0.262–2.84 m (Keith et al., 2000) and 0.01–6.45 m (Dean et al., 2003; Dean and Roxburgh, 2006) respectively.

The relationships between biomass and stem volume, for these three eucalypts, are known within limited ranges of DBH (mentioned in Chapter 1). Their stem volumes can be calculated using the Farm Forestry Toolbox (FFT) (Warner, 2007), which requires input of DBH and tree height. Within the FFT the majority of stem taper formulas and their accommodated range of DBH, for species encountered in this study, were commercial-in-confidence and consequently they could not be deployed in this project. Only one of the formulas was publically available— for *E. obliqua* (Goodwin, 1992), which accommodated DBH 0.1–2.8 m. Nevertheless, one useful observation from using the FFT, was that for a given DBH and height, stem volume was in the order wet *E. obliqua* > *E. delegatensis* > *E. regnans*. Biomass allometrics based on DBH alone for the three species are compared in Keith et al. (2000) over

the DBH range 0.1–1.0 m: the same sequence applies to biomass, as noted for stem volume from the FFT. Thus the sequence represents common trends in both taper and basic density. Thus combining *E. obliqua* and *E. regnans* allometrics would have provided a reliable approximation for *E. delegatensis*.

The allometric for merchantable volume could not be directly converted to one for biomass owing to the location-specific criteria for merchantability and the absence of an accompanying allometric for entire stem volume. However in the same report there was a merchantable volume allometric for *E. regnans* over a comparable DBH range (Wang and Hamilton, 2003). A new allometric equation was derived using the approximation that merchantable volume was an equal fraction of the whole stem volume for both species. When combined with the allometric for *E. regnans* stem volume from Dean and Roxburgh (2006) the ratio of merchantable volumes for the two species yielded the stem volume of *E. delegatensis*, for a given DBH. The stem volume was converted to dry biomass using the basic density of 524 kg m^{-3} for *E. delegatensis* (Ilic 1997) and combined with other aboveground components (e.g. branches and leaves) as for *E. regnans* from Dean et al. (2003), to provide an allometric for aboveground biomass for *E. delegatensis*. The corresponding allometric was called *Edel_by_ratio*.

Thus there were three possible biomass allometrics for *E. delegatensis*: *E. obliqua*, *E. regnans*, and *Edel_by_ratio*. As none of the three was overwhelmingly more suited to *E. delegatensis* over the sampled range of DBH, the average of the three, forming a new allometric equation, was used. That average was fitted by [non-linear] regression in Minitab 17 statistical software to a logistic form. Settings in Minitab were left at defaults (apart from the choice of equation) and the refinement algorithm used was Gauss-Newton. A range of equation types was examined using the computer program Eureqa which performs [automated] symbolic regression (Schmidt and Lipson, 2009), but the fit was only improved on the logistic form when using an order five polynomial (i.e. six parameters instead of the three for the logistic function). The logistic function was therefore retained, and it had the additional benefits of being better defined than the polynomial for high DBH (similarly when

compared with log/log allometrics, which can increase exponentially) and providing reasonable biomass for DBH below 0.1 m. Adoption of the logistic form had also been found suitable for juvenile to advanced-mature *E. regnans* allometry (e.g. Dean et al., 2003; Dean and Roxburgh, 2006).

3.3.3.2 Rainforest understorey allometric

For the rainforest understorey species, no species-specific biomass allometrics were available. For only one of the observed species (*Acacia dealbata*), stem volume could be calculated using the FFT, however the formulae were not available, the applicable DBH range in the FFT was unknown, and tree heights had not been measured. Species-specific allometrics are unavailable for many of the understorey species, including that with most likely the highest biomass, myrtle (personal communication, Forestry Tasmania, 2009). Two generic biomass allometrics for rainforest species were: (i) for temperate rainforest species (Keith et al., 2000) and (ii) wet-sclerophyll and mixed-forest species (Dean et al., 2003). The Keith et al. (2000) allometric was comprised of two parts: generic rainforest, and a correction for temperate rainforest (in the form of graphed data points), both covering the DBH range 0.1–1.0 m. A single allometric for temperate rainforest was derived from those two components[4]. The temperate rainforest allometric gave an aboveground biomass for a rainforest tree of DBH 2 m that was 131% of that for an *E. delegatensis* tree of the same DBH (using the average allometric for *E. delegatensis* derived in this work— Eq3-2 below). This was considered unreasonable because of the height difference between the two species, namely canopy versus understorey, and because a myrtle of that size in the mixed-forests of the Styx and Florentine Valleys would most likely have a substantial portion of senescence. The allometric was adjusted for

[4] That derivation as shown in the original publication of this work— Dean et al. (2012) doi:10.1080/11263504.2011.638332, was found to be incorrect (Barrie May, CO_2 Australia, personal communication, 2012) and it was corrected in the present work.

senescence by subtracting 33.33% for trees with DBH≥ 1.5 m and using Eureqa (Schmidt and Lipson, 2009) to form a new, single allometric for the DBH range 0.1–3.0 m. The 33.33% corresponds to the second of three decomposition stages used for coarse woody debris (1, 0.3333 and 0.6666 of original mass present).

The allometric given in Dean et al. (2003), was intended for use within the spatio-temporal carbon modelling software, CAR4D. In applications of CAR4D however the formula was rarely used in its raw form, but adjusted to suit environmental conditions through scaling the biomass magnitude and growth rate (Dean et al., 2004). The most common adjustment for mixed-forest was to scale the understorey magnitude by 0.5 (e.g. Dean et al., 2004; Dean and Roxburgh, 2006) ('Dean et al. 2003/4' hereafter). That adjustment was applied here. Thus there were two allometrics available for the rainforest species: (i) temperate rainforest extrapolated and adjusted for senescence, and (ii) Dean et al. 2003/2004.

As neither of the Keith et al. (2000) temperate-rainforest-corrected-for-senescence allometric, nor the Dean et al. 2003/4 allometric were overwhelmingly more suited to the understorey in *E. delegatensis* mixed-forest over the sampled range of DBH, the average of the two was used (which provided an additional gauge to uncertainty in understorey biomass). That average was formulated as a quadratic-logistic function, fitted by symbolic regression in Eureqa. As for *E. delegatensis*, symbolic regression revealed that the fit could only be improved using a polynomial of order 5, and therefore the logistic function was retained.

DBH size distributions of species (per hectare) for the plots were tallied across the study area, corrected for plot size (in projection), allocated to DBH classes and graphed as histograms. DBH class width was varied between species to provide maximum clarity in the histograms, (e.g. class width for *E. delegatensis* was 0.2 m). The aboveground biomass allometrics were applied to the categories, rather than to the trees separately prior to tallying. The C in aboveground biomass per unit area for *E. delegatensis*, in the sample plots, was calculated using the three allometrics, and averaged; the average was equivalent to using the derived logistic function. The C in

biomass for *E. delegatensis* was multiplied by 0.7, 0.75, 0.8 and 1 to test the effect of accommodating senescence on stand-level carbon stocks (i.e. the sensitivity of the allometrics to senescence). Similarly to *E. delegatensis*, the C in aboveground understorey biomass was calculated for both contributing allometrics and their average, for comparison purposes.

The allometric used for determining the carbon in snags was the average of the three eucalypt allometrics described above for aboveground biomass, i.e. the penultimate allometrics for *E. delegatensis*. Apart from height loss, to accommodate general loss of branches, bark, and unrecorded internal voids, the calculated aboveground mass was multiplied by 0.6666, in addition to the recorded decay class factor. For example, a snag of decay class S (i.e. 0.33333) had the necromass of an equivalent-sized living tree, multiplied by 0.2222. Loss of height was interpreted by multiplying the necromass from the allometric by the proportion of the original height remaining. That, in effect, assumed a cylindrical stem and was therefore more conservative than volume based on stem taper, when more of the upper stem had decomposed. The original height was estimated from the height-DBH relationship for *E. regnans* (Dean et al., 2003) but with an adjustment to accommodate *E. delegatensis* possibly being of shorter stature, and to blend with the measured DBH-height pairs; i.e. a new allometric was derived. Parameters in that height allometric were varied to gauge its impact on calculated necromass. The volume of hollows, fallen logs and branches was calculated as a frustum of a cone. For snags and stumps the volume was converted to mass using the basic density of 524 kg m^{-3} for *E. delegatensis* (Ilic, 1997) and applying the same decay factors as for the solid section (i.e. all necromass was calculated as if it were *E. delegatensis*.) For fallen logs and branches the volume was converted to mass using a conservative basic density of 400 kg m^{-3}, and the recorded decay class. Necromass was pooled and calculated on a per plot basis, then averaged over the study area.

3.3.4 State-wide, wet-eucalypt data

Spatio-temporal, electronic records of forestry operations in wet-sclerophyll and mixed- *E. delegatensis* forests from 1999–2009 were obtained from the Forest Practices Authority (FPA) (the limit of electronic records at the time of the project). The data were examined for trends in areas logged, and silvicultural prescriptions over that period, for the forest type in the study area.

Potential biomasses for wet- *E. delegatensis*, *E. obliqua* and *E. regnans* forests, can be calculated from the continental potential biomass layers from Berry & Roderick (2006) (termed B&R here onwards) and FullCAM (Richards and Brack, 2004), by overlaying them (using GIS) on the vegetation community layer (TASVEG, DPIW 2010). The FullCAM and B&R potential biomass layers were derived by remote sensing and ground-truthing (i.e. maturing and mature vegetation) but with minimal calibration for mature wet-eucalypt (i.e. higher biomass) forests, resulting in the biomass being underestimated for that forest type (Richards and Brack, 2004; Berry and Roderick, 2006; Berry, 2010; Keith et al., 2010). Thus those layers are unsuitable for calculation of absolute values, for mature wet-eucalypts in Tasmania but nevertheless, they are satisfactory for ranking relative potential biomass of similar wet-eucalypt forest types within a region. They were used to compare relative biomass for the three major ash wet-eucalypt forests in Tasmania, and to compare the potential biomass at the study site against that for the average wet-eucalypt *E. delegatensis* forest in Tasmania.

The B&R layer quantified C in biomass (including roots), whereas the FullCAM layer included only C in aboveground biomass. To average the two layers, 15% (representing root biomass) was subtracted from the B&R layer before averaging with that from FullCAM. (The figure of 15% was an average of 10.5% for ash eucalypts (Feller, 1980) and 20% for temperate, high-biomass eucalypt forests (Mokany et al., 2006).) The B&R and FullCAM layers did not have common medians across the range of vegetation communities in Tasmania, i.e. the two layers were effectively on different scales. Therefore, to compare the relative potential

biomass for forests dominated by the three eucalypts used in allometrics here, and to employ data from both layers, the potential biomass for each vegetation community in Tasmania was scaled from 0–1 (normalised), separately for the two layers. Once normalised, the average between the two layers for each forest-type was calculated, to yield a single ranking scheme.

3.4 Results

3.4.1 Allometrics

Comparison of existing stem volume allometrics for *E. obliqua*, *E. delegatensis* and *E. regnans* (Figure 3-5) showed that extrapolation of the allometric for *E. delegatensis* beyond its range to accommodate mature trees would have produced unreasonably high volumes (Figure 3-5). The stem volume of *E. delegatensis*, from the ratio of merchantable volumes of *E. delegatensis* and *E. regnans* derived from the allometrics reported in Wang & Hamilton (2003) was:

$$Edel\_by\_ratio = Ereg\_vol \times \left(0.62400 + \left(3.6428 \exp\left(-0.3337 DBH^{0.57072}\right)\right)\right)$$

Eq3-1

where: *Edel_by_ratio* is the stem volume of *E. delegatensis* in m^3, *Ereg_vol* is the stem volume of *E. regnans* in m^3 from Dean and Roxburgh (2006), and DBH is in metres. The stem volumes from Eq3-1 ('Edel_by_ratio') were higher than for *E. regnans* at small DBH (e.g. *E. delegatensis* volume was twice that of *E. regnans* when DBH 0.08 m), but the ratio decreased to 1 when DBH 0.285 m, after which it yielded *E delegatensis* volumes less than those of *E. regnans* (Figure 3-5).

The averaging of the allometrics for *E. obliqua*, *E. regnans* and *Edel_by_ratio* produced volumes that were below that for *E. regnans* for DBH from 0.15–3.3 m but above, thereafter (Figure 3-5). It also generally fitted the sequence of stem volumes

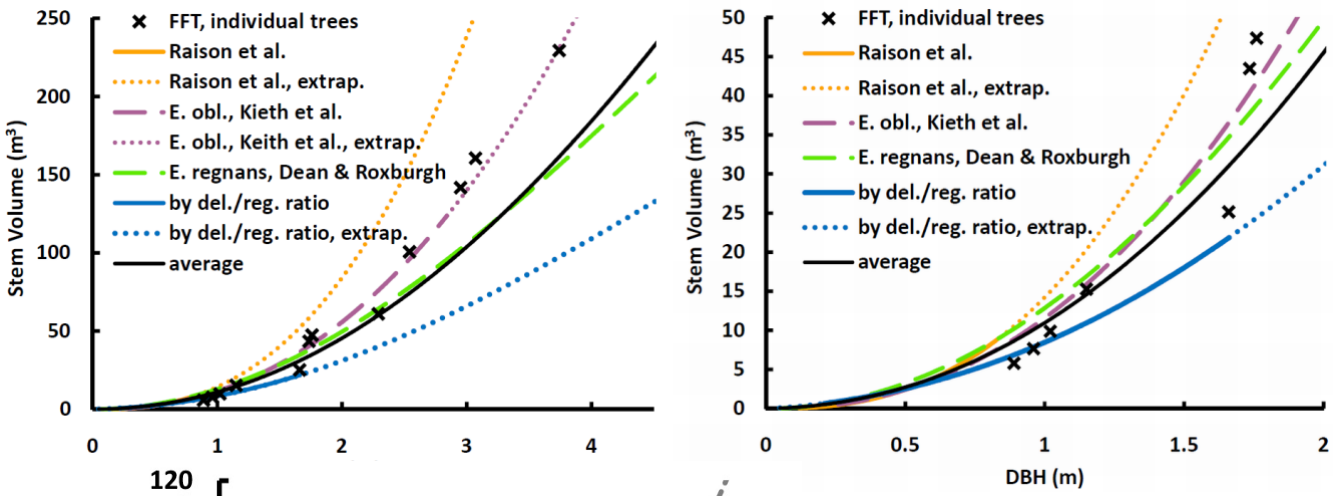


Figure 3-5. Stem volume versus DBH for different allometrics tested for E. delegatensis. Left: DBH range on the study site Right: close-up at low DBH. Dotted lines: allometric from literature extrapolated beyond its original DBH range. Crosses show volumes calculated using FFT, for individual trees which had their DBH and height measured (DBH range accommodated in FFT is unknown).

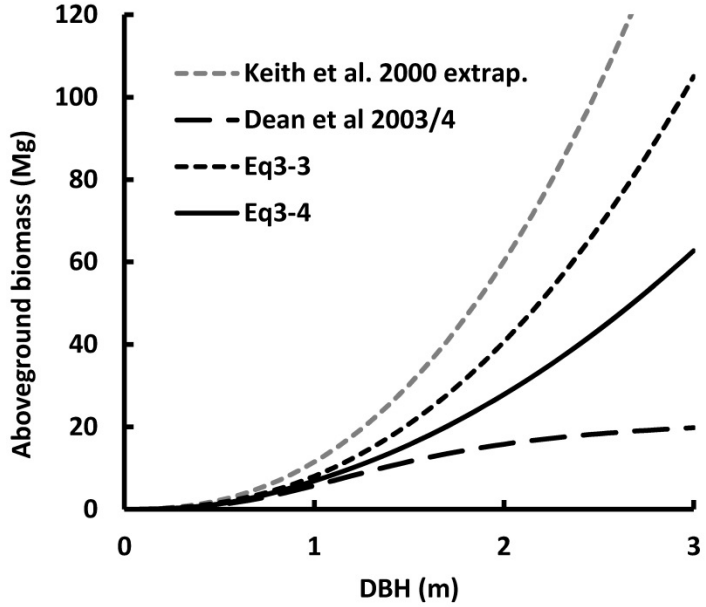


Figure 3-6. Allometrics for carbon in aboveground biomass, for rainforest understorey species. Keith et al. 2000 extrap.' corresponds to the rainforest allometric of Keith et al. (2000) with the temperate rainforest correction applied, and extrapolated beyond the 1 m DBH limit of the original work.

calculated for the species from the FFT but indicated lower biomass for higher DBH (i.e. it was more conservative). The logistic function for *E. delegatensis* aboveground biomass, corresponding to the average of the three contributing allometrics (*E. obliqua*, *E. regnans* and *Edel_by_ratio*) was:

$$Edel\_AGB = 1612.4 \times \left(1 - \left(\frac{1}{1 + \left(\left(DBH + 0.01\right)/12.714\right)^{2.2283}}\right)\right)$$

Eq3-2

where *Edel_AGB* is in Mg of dry matter, and DBH is in metres (N= 800 modelled points, DBH=0.01–8.0 m). The forecast biomasses from Eq3-2 (i.e. the average of the three allometrics) were generally between those of *E. obliqua* and *Edel_by_ratio* and close to those for *E. regnans*, for most values of DBH (similar to the corresponding relationships for stem volumes, Figure 3-5).

Combining the biomass allometric for generic rainforest species with the correction points for temperate rainforest species from Keith et al. (2000) adjusted for senescence at high DBH, gave the following biomass allometric for temperate rainforest species:

$$Temp\_rain\_AGB = 8.02758 DBH^{2..3401}$$

Eq3-3

where *Temp_rain_AGB* is aboveground dry biomass in Mg, and DBH is in metres. The average of the two allometrics (Eq3-3 and Dean2003/4) for understorey biomass could be represented by the following quadratic-logistic function:

$$und\_ave = 6.9683 DBH^2 \left\{1/\left[1 + \exp\left(-4.37497 DBH^2\right)\right]\right\}$$

Eq3-4

where *und_ave* is aboveground dry biomass in Mg, and DBH is in metres (N= 300 modelled points, DBH=0.01–3.0 m). Eq3-4 (i.e. the average of the two allometrics,

Figure 3-6) produced biomasses in between the two averaged allometrics (as expected) and with a lower biomass than for *E. delegatensis* for high values of DBH (e.g. 2 m).

3.4.2 Stand-level distributions

The stand density of rainforest understorey outnumbered that of the eucalypt canopy by around fourfold (Table 3-1). Conversely, average DBH of the eucalypts was fivefold that of the understorey. Apart from annual growth increments and longevity, that is related to the continual regeneration of the myrtles and sassafras without fire (Read & Hill 1988, Ashton 2000), which was indicated in the stand-density-DBH histograms, showing marked positive skew (Figure 3-7). Celery-top pine is more likely to require fire for regeneration than do myrtle or sassafras— accordingly the smallest specimens were noted in the same plot where a eucalypts had notable fire scars. A single age-cohort of pole-stage or older ash eucalypts usually has a DBH frequency distribution in the shape of a log-normal distribution, i.e. showing a positive skew (e.g. Ashton 2000; Ashton, personal communication, 2002). The first peak in the DBH-frequency histogram was at the minimum recorded DBH and there was a second where that youngest cohort overlapped the next-oldest cohort (near DBH=1 m) (Figure 3-7), which suggests that there were at least two age cohorts of *E. delegatensis* present.

Table 3-1. Numbers of species present and primary DBH statistics. Standard deviation in brackets. N= number of trees measured.

Species	N	DBH (m)		
		min.	max.	average
E. delegatensis	207	0.21	4.50	1.63(0.81)
understorey total	897	0.01	2.52	0.31(0.29)
myrtle	353	0.01	2.52	0.41(0.37)
sassafras	332	0.01	2.05	0.30(0.20)
celery-top pine	83	0.01	0.51	0.17(0.13)

The *E. delegatensis* DBH frequency distribution (which was a compound of separate cohorts) showed a positive skew (sample skewness 1.6) (Figure 3-7) but *E. delegatensis* DBH carbon distributions (Figure 3-8) showed a negative skew (for the allometric average, sample skewness was -0.41). That negative skew was more pronounced using the extrapolated allometric for *E. obliqua* (sample skewness -1.3)— corresponding to its much higher biomass for high DBH when compared to the other allometrics (Figure 3-5). With the derived biomass allometric (Eq3-2, i.e. the average of the three contributing allometrics) being conservative, it is noteworthy that the relatively small percentage of high DBH trees (~1.1 tree per hectare) from DBH 3.5–4.5 m contributed ~16% to the stand-level, *E. delegatensis* biomass. The contribution of high DBH trees is also reflected in the DBH cut-off that represents 50% of the carbon distribution, being at 2.4(0.1) metres (Table 3-1). That DBH value did not significantly alter with variation of assumed senescence.

The effect of eucalypt senescence on the DBH carbon distribution was only marginally noticeable in the histograms and was probably more related to the difference between plots. For example the maximum difference (between with- and without senescence) for any DBH class was ~13%, (between DBH classes 2.1 and 2.9 m), but it was only 3% for the DBH class of 2.7 m. Overall, the effect of attributing 30% missing biomass to senescent trees only altered the stand-level *E. delegatensis* biomass by ~8% (e.g. from the average of the three allometrics, Table 2).

The DBH carbon distributions for understorey species, when pooled together, showed a marked positively-skewed distribution (Figure 3-9). However, for celery-top pine and sassafras, the higher DBH classes (0.525 m and 2.05 m respectively) contributed more to the stand-level carbon stocks than did their minimum DBH classes, even though those upper DBH classes were not near the upper limit of the Dean et al. 2003/2004 allometric.

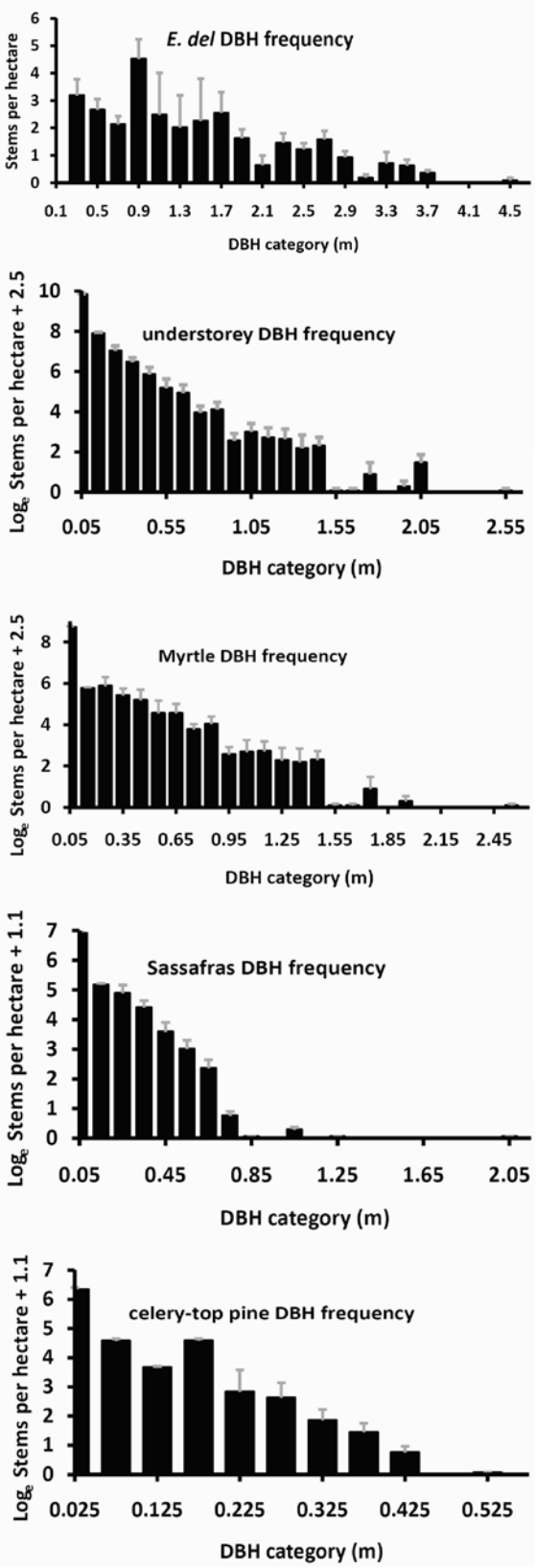


Figure 3-7. DBH frequency histograms. Those for understorey species were converted to natural logarithms (on the ordinate) for clarity. Where the number of stems per hectare in any DBH class was less than 1 then the ordinate was shifted upwards for clarity (factor noted on axis label). Error bars (positive) are the standard errors corresponding to variability between the 10 sample plots.

The understorey contributed 144(21) Mg ha^{-1} (27%) to the stand-level carbon in biomass, with the majority of that (62%), coming from myrtle (Table 3-3). The contribution of only 1% from celery-top pine was due to its limited occurrence and smaller average size.

For *E. delegatensis* the 25%-senescence scenario was considered most likely and sufficiently conservative for this site. Using the carbon in biomass for that scenario from the *E. delegatensis* allometric-average (340(104) Mg ha^{-1}) (Table 3-2), and the understorey carbon from the understorey allometric-average (143(21) Mg ha^{-1}) (Table 3-3), gave the stand-level carbon in biomass of 483(124) Mg ha^{-1}.

Table 3-2. Effect of different allometrics and % senescence on C stock for *E. delegatensis*. Also showing DBH category above which lies 50% of stand-level carbon. The allometrics used for 'Eobl_equiv' and for 'Ereg_equiv' from Keith et al. 2000 (extrapolated), Dean et al. 2003, and Dean & Roxburgh 2006. Standard deviations in brackets are from the average of the three allometrics.

	Allometric				
	Eobl_equiv	Ereg_equiv	Edel_by_ratio	Average	
	Stand-level C (Mg ha^{-1})				% of total stand biomass
0% senescence	459	382	242	361(110)	75
20% senescence	438	364	231	344(105)	71
25% senescence	432	359	228	340(104)	70
30% senescence	427	355	225	336(103)	69
DBH cut-off for 50% of stand C (m)	2.54	2.39	2.37	2.43(0.09)	

Table 3-3. Carbon in aboveground biomass for understorey species, from allometrics.
Also, DBH category above- which is 50% of stand-level carbon. 'Keith et al. (2000) extrapolated' is Eq3-3. Standard deviations in brackets are from the average of the two allometrics.

	Allometric				
	Keith et al. 2000. extrapolated	Dean et al. 2003/2004	Average	%understorey	% of total stand biomass
	Stand-level C (Mg ha^{-1})				
all understorey	177	109	144(21)	100	27
myrtle	107	69	88(18)	62	18
sassafras	54	32	43(5)	30	8.9
celery-top pine	6.2	3.4	4.8(0.7)	3.4	1.0
	DBH cut-off for 50% of stand C (m)				
all understorey	0.60	0.60	0.60(0.01)		
myrtle	0.79	0.77	0.78(0.01)		
sassafras	0.44	0.45	0.44(0.01)		
celery-top pine	0.20	0.23	0.21(0.02)		

Variation of parameters, within realistic ranges, for the height allometric for snags only contributed to ~5% variation in the total C in necromass of 153(51) Mg ha^{-1}, due to the relatively small contribution from snags (Table 3-4). Combining the stand-level biomass carbon with the necromass carbon, gave a stand-level aboveground carbon of 636(172) Mg ha^{-1}. Thus the understorey contributed ~23% to the total, and necromass contributed ~24%, of which 83(13)% was from fallen logs, i.e. fallen logs contributed ~21% to aboveground carbon, which was roughly equivalent to the ~23% contribution from the understorey species.

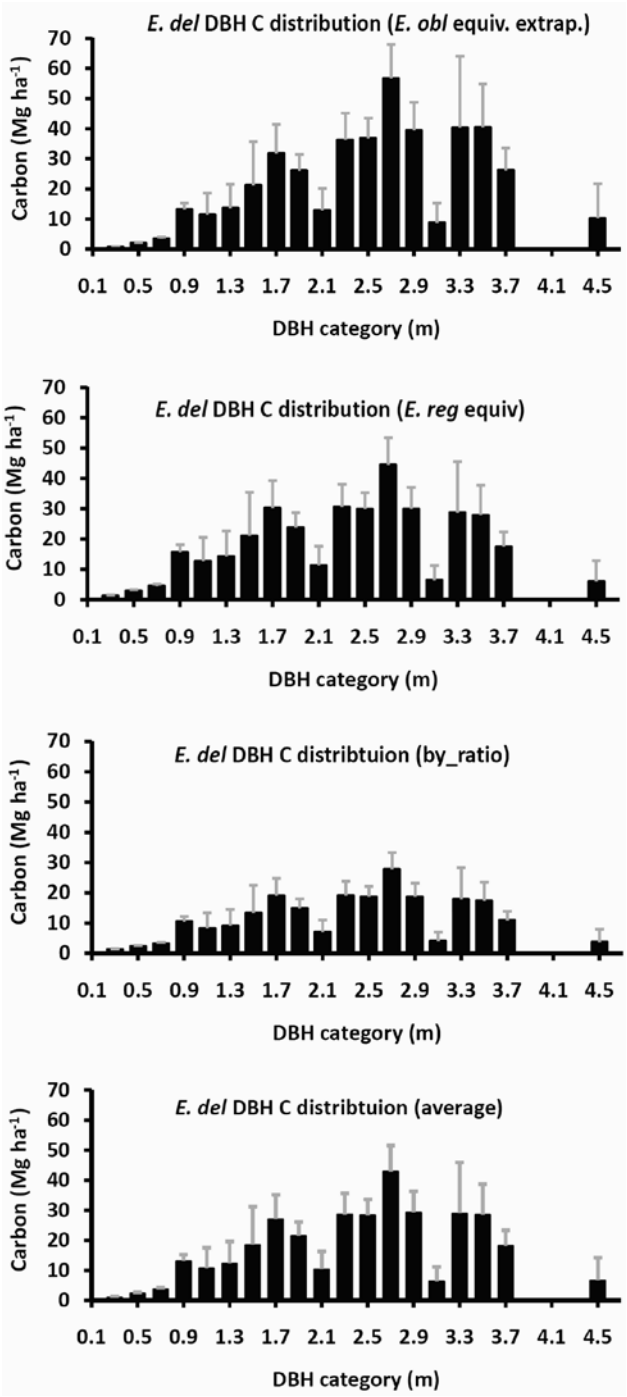


Figure 3-8. DBH carbon histograms for *E. delegatensis* using different allometrics. Error bars (positive) are the [fractional] standard errors corresponding to variability between the 10 sample plots [multiplied by the category values].

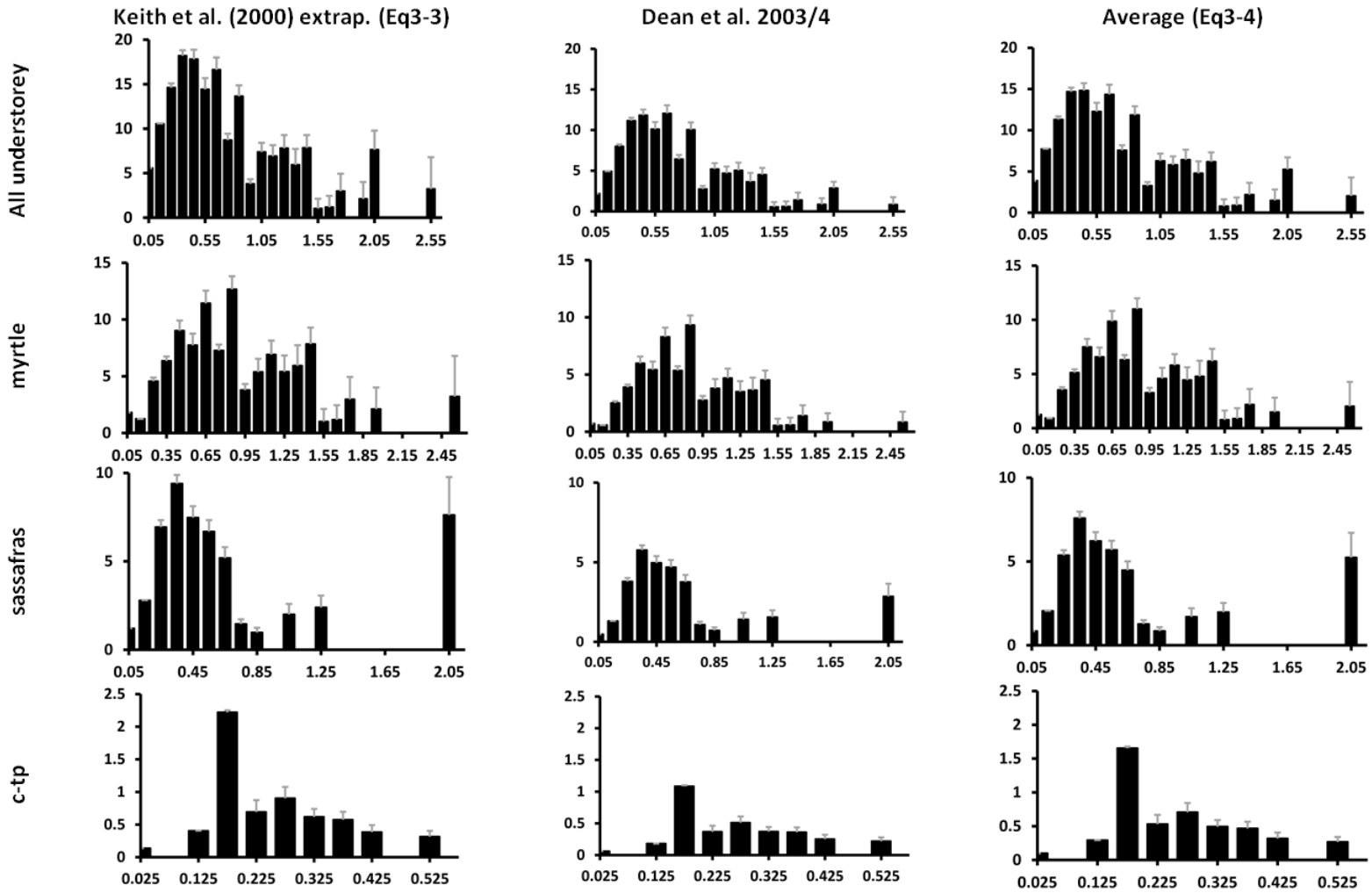


Figure 3-9. DBH carbon histograms for rainforest understorey using different allometrics.
'c-tp': celery-top pine. Error bars (positive) are the [fractional] standard errors corresponding to variability between the 10 sample plots [multiplied by the category values]. Abscissa: DBH category in metres. Ordinate: carbon stock in Mg ha^{-1}.

Table 3-4. Carbon in necromass (within the 30 m radius circle) summed per plot. Separated into snags and fallen logs. (St. Dev. in brackets)

Plot	Snags	Fallen logs	Necromass	Fallen logs % of total
		(Mg ha^{-1})		
1	26.8	141.5	168.3	84.1
2	2.5	196.0	198.5	98.7
3	10.4	124.0	134.4	92.3
4	83.9	142.0	225.9	62.9
5	13.8	91.4	105.2	86.9
6	43.7	83.8	127.4	65.7
7	9.2	75.5	84.7	89.1
8	7.3	215.0	222.3	96.7
9	32.2	65.8	98.0	67.2
10	16.1	149.9	166.1	90.3
Average	25(24)	128(51)	153(51)	83(13)

3.4.3 State-wide data and logging emissions

The average carbon in biomass for the 10 plots, as derived by GIS analysis from the FullCAM & B&R layers, was 69.5(0.4) and 115.5 Mg ha^{-1} respectively, which was within one standard deviation of the State-wide averages of 66(37) and 101(15) Mg ha^{-1} (Table 3-2). Thus the site was reasonably representative for potential carbon density in that forest type, State-wide. Using the 50% long-term decline in carbon stocks noted for conversion of primary-forests to a harvesting cycle in Chapter 2, if the forest in the study area underwent the same LUC then the forecasted, long-term aboveground-carbon emission is 311(90) Mg ha^{-1}.

The logging records from the FPA revealed that conversion of wet *E. delegatensis* forest to plantation had declined overall from 1999–2009 at a rate of ~89 ha year^{-1} (R^2=0.42), however 'clearfell, burn and sow' (sowing of eucalypt seed) (CBS)-type logging showed a slight increase (~28 ha year^{-1}, R^2=0.36, P= 0.053). The total area of wet *E. delegatensis* forest logged in that period was 38,225 ha (Table 3-5), of

which 7920 ha was converted to plantation and 5523 ha was converted through CBS. The long-term aboveground-carbon emission from that clearfelling, using the 50% drop, and the long-term emission of 311(90) Mg ha^{-1}, is likely to be as high as 4.2(1.2) Tg, depending on the growth rate and maturity-stage of the primary forest. Additionally, from State-wide inventory, Moroni et al. (2010) found 232 Mg ha^{-1} for mature (i.e. ≥110 years) wet-eucalypt forests (averaged over all eucalypt types, mixed-forest and wet-sclerophyll forest). However their carbon density for mature forest was not significantly different to their value for average-aged forest, and there was no differentiation of carbon densities between late-mature and early-mature primary forest even though the former can be 50% higher (Figure 2-8; Appendix-IV.6). Nevertheless, assuming their value was applicable to late-mature, primary, wet *E. delegatensis* forest then the long-term carbon emission (50% of stock) would be 1.6 Tg. An average of the carbon density (which appears representative of potential biomass in *E. delegatensis* State-wide) with that from Moroni et al. (2010), may well be representative of the growth and maturity stages of the primary forests logged from 1999–2009: giving a long-term aboveground-carbon emission of 1.6–4.2 Tg, i.e. 2.9(±1.3) Tg.

Table 3-5. Acreages of main wet-eucalypt types logged from 1999–2009 and average potential carbon in biomass derived by overlaying the FullCAM and B&R layers on vegetation mapping (TASVEG). Values were normalised (State-wide, 0 to 1 for the two layers) then averaged between the two layers, to allow comparison between forest types.

wet-eucalypt species	Area logged (ha)	Area remaining (ha)	Number of TASVEG polygons (thousands)	C in biomass (Mg ha^{-1})		
				FullCAM	B&R	Normalised & Averaged
E. delegatensis	38224	276378	4135	66(37)	101(15)	0.59
E. obliqua	60384	464082	7425	84(53)	106(24)	0.73
E. regnans	20485	76891	1230	82(53)	117(20)	0.76

3.5 Discussion

3.5.1 Allometric derivation

The allometrics derived were stable over all of the DBH values measured in the field, i.e. they did not increase exponentially at high DBH and did not produce negative values for DBH <0.1 m. The *Edel_by_ratio* may have produced *E. delegatensis* values of biomass too high when DBH was below 0.29 m, but that effect would have been cancelled to some degree by averaging with the other two allometrics. Additionally, when the *E. delegatensis* biomass for the study site was tallied then high biomass at low DBH values was not observed (Figure 3-8). For DBH>0.53 m, stem volume from *Edel_by_ratio* was the smallest of all allometrics, which concords with the more-pronounced buttressing of *E. regnans* (which tapers faster and therefore has lower stem volume for the same DBH). For the same reason, the similarity of biomass calculated for *E. delegatensis* (from the average allometric, Eq3-2), and for *E. regnans*, indicates that Eq3-2 (i.e. the average of the three allometrics) was conservative. That was confirmed by comparison with the relative volumes and biomasses for the three eucalypt species described above, and by the volumes from the FFT being higher (for high DBH) than the average allometric. Thus it is unlikely that *E. delegatensis* biomass was overestimated in the present study.

The allometric derived for the understorey in the mixed-forest, being of quadratic-logistic form, was similarly stable at high values of DBH. Until destructive sampling is undertaken, such as during logging operations to minimise C footprint of measurement (e.g. Ximenes et al., 2008a), or accurate taper formulas for the rainforest species become publically available, it is difficult to increase the accuracy of C stock estimates. Although an average of the allometrics for mixed-forest (Dean et al., 2003; Dean et al., 2004) and temperate rainforest (Eq3-3) was derived (Eq3-4) and applied here, there was no compelling reason to assume its effect was more accurate than that from the original mixed-forest allometric alone. That would

decrease the carbon contribution from understorey species by 24%, and reduce the total aboveground carbon by 3% to 602 (174) Mg ha^{-1}.

3.5.2 Carbon distribution and significance

Three factors indicating the importance of reliable allometrics for high DBH eucalypts, from the present study are: (i) the negative skew on the DBH carbon distribution, (ii) the 50% DBH cut-off of 2.4(0.1) m for *E. delegatensis*, and (iii) the 16% contribution to eucalypt carbon stocks from the ~1.1 trees ha^{-1} with DBH≥3.5 m. The number of individual trees measured (over a thousand) through the deployment of citizen science was high. Similarly the area measured (~7.7 ha) is amongst the highest measured with a methodology suitable for carbon assessment of TOF; higher than the 3.2 ha of Keith et al. (2009) and Fedrigo et al. (2014). If the study site had comprised only a single age cohort of eucalypts then the high DBH trees would have contributed a marginally higher percentage. The result of the relatively few larger trees representing a substantial portion of the carbon is therefore noteworthy, although of course, as always, more data would have been better, but it may have included even larger trees. The *E. obliqua* biomass allometrics can accommodate up to DBH 2.8 m, but that is the highest in the review of allometrics in Keith et al. (2000). The *E. regnans* allometrics (Dean et al., 2003; Dean and Roxburgh, 2006) were calibrated with trees up to ~6.44 m DBH and corrected for cross-sectional area deficit. Thus, in general, more work is required on C accounting of late-mature, primary, wet-eucalypt forests. The higher standard error on the highest DBH category in the carbon-DBH histograms of some of the understorey trees reflects the more-sporadic location of such large trees.

The necromass contribution of 24% to aboveground carbon is above the mean value for the wider category of Australian tall-open forests (Woldendorp and Keenan, 2005). The absolute value of the stock however, agrees with other values for Tasmania, which increase with more southern latitudes (i.e. cooler temperatures),

and the proportion of necromass in fallen logs agrees with higher values noted in old in oldgrowth forests (Woldendorp and Keenan, 2005).

The high, measured-biomass compared with the potential value in the FullCAM and B&R layers concurs with that found previously for primary wet-eucalypt forest, in Chapter 2. The current, wide use of the FullCAM layer suggests that analysis of primary-forest C stocks can provide useful calibration points for its adjustment.

The 483(124) Mg $ha^{-1}$of aboveground biomass carbon found here, is within the range of values for mature primary-forests measured elsewhere, and just above the median value for temperate forests in the review of Keith et al. (2009). For example, it is above the 160–268 Mg ha^{-1} in the Carpathian mountains (Szwagrzyk and Gazda, 2007; Keeton et al., 2010) but lower than the 642–723[5] Mg ha^{-1} elsewhere in Tasmania (Dean and Roxburgh, 2006), lower than the 843–1819 Mg ha^{-1} in Victoria (Australia) (Van Pelt et al., 2004; Keith et al., 2009), and lower than the 647 Mg ha^{-1} in the USA for *Pseudotsuga menziesii* Mirb. (Franco)) (Van Pelt et al., 2004). Nevertheless, the aboveground biomass carbon measured in this work, when compared with two other examples of Tasmanian primary-forest (Dean and Roxburgh, 2006), aligns well with the sequence of normalised Tasmanian State-wide potential biomass layers (Table 3-5): i.e. 466<642<723 Mg ha^{-1} corresponding to 0.59<0.73<0.76 for *E. delegatensis*, *E. obliqua* and *E. regnans* respectively.

Aboveground carbon, one of the major carbon pools, is the easiest pool to measure in native forests. These carbon stocks in mature, primary wet-eucalypt forests, can be much higher than those used in routine accounting (e.g. Keith et al., 2009). With the landscape-level, age-averaged stock of carbon being the pre-logging benchmark, it is necessary to account for such potentials. Upper stock limits are likely when the forest has marked oldgrowth characteristics, with other components that may be either younger age classes of the canopy species (recruits resulting from non-stand-

[5] Although the value for C in aboveground biomass reported in Wood et al. (2010) for the Tasmanian *E. regnans* was 850 Mg ha^{-1} the value mentioned above in this paragraph is 723 Mg ha^{-1}, as the former actually included root carbon.

replacing fires or non-fire-induced recruits) or rainforest understorey species capable of continuous regeneration in the absence of fire. The data collection and processing presented here provides a clear example of how the pre-logging stocks can be tallied for use in regional comparisons.

Chapter 4 Are there any circumstances in which logging primary wet-eucalypt forest will not add to the global carbon burden?

This chapter is adapted from a published paper: Dean et al. (2012d)

4.1 Abstract

The major influences of commercial forestry initiated over recent decades on Tasmanian primary-forest carbon are identified, and the means to mitigate their ongoing impacts are investigated. Spatio-temporal trends were derived from records of commercial operations combined with biomass data. Over the last two decades, the majority of forest C destined for short- or long-term emission (LTE, i.e. over several centuries and multiple harvests) was from clearfelling the higher-biomass wet-eucalypt forests on public land. Carbon dynamics at the unit-area-level for logging two disparate, wet-eucalypt forests were modelled. Parameters were varied to determine management options and model sensitivities under conversion by clearfell and intense burn to either eucalypt plantation or forest regeneration from local eucalypt seed. The first cycle of conversion of primary forests contributed 43(±5)% to the LTE, and the LTE constituted ~50% of the primary-forest C stock. Whether the first logging of even-aged primary forests was prior to or after maturity, the LTEs were equivalent, although short-term emissions were ~2 x higher from oldgrowth. Minor variations in soil organic carbon efflux during operations significantly altered LTEs. Conversion of wet-eucalypt by clearfell from 1999 to 2009 incurred an LTE of 2(±1.6) Tg from each year's logging. Lengthening the harvesting interval for sown forests from 80 to 200 years reduced LTEs by 42%, and eucalypt wood-products by 26%; but yielded 40(±20) Mg ha^{-1} of C in rainforest—helping to sustain mixed-forest ecosystems and their products. Using 200-year cycles

for the wet-eucalypt already clearfelled could avoid LTEs of ~15 Tg. Long-term C dynamics under harvest cycles were constrained by mathematical precepts that facilitate climate change modelling, e.g. the time to reach the harvesting-cycle's asymptote is correlated to the half-life of the longer-lived C pool. Emissions are not recovered by sequestration in wood-products unless their half-lives are ~10x contemporary values— requiring 200–1000 years for recovery, during which time emissions would augment global climate change. Emissions can be reduced by product substitution, and by recycling wood-products, in a stable wood market (i.e. without 'leakage', where alternative forest is logged). Primary-forest is part of a global commons. Comprehensive C accounting cannot occur if logging effects are omitted.

4.2 Introduction

Improved climate change modelling could yield economic benefits for industries (Durant et al., 2011), and improved C accounting for the forestry sector, which must be implemented within the next 5 to 7 years, to meet treaty and policy obligations (Law and Harmon, 2011). Australia's forest industry is reported to be a net C sink (Department of the Environment, 2016), when accounting is only for the [Kyoto-compliant (post-1990), Article 3.3] plantation estate, reforestation and afforestation. Also, the native forest sector has been reported to be a net C sink (MPIGA, 2008). The calculations on which that conclusion was based included biomass sequestration for recovery from earlier logging but excluded emissions from wood-products extracted prior to 1971, effects on soil and coarse woody debris, and non-target species. MPIGA (2008) underestimated the primary-forest C for the high-biomass forests in south-eastern Australia (Keith et al., 2010). Partial C accounting such as that of MPIGA (2008) can discourage climate change mitigation by suggesting an industry C balance which does not exist (Shvidenko et al., 2010; Law and Harmon, 2011). Logging history is vitally important in calculating the carbon implications of forestry, as a greenhouse benefit from carbon in wood-products generally starts only

after several cycles of harvesting secondary forests, whereas usage of wood-products from primary-forests (precipitating the first logging cycle) constitutes a net emission (e.g. Harmon et al., 1990; Sathre and O'Connor, 2010; Fix et al., 2011; McKinley et al., 2011).

Long-term emissions (LTEs) (i.e. ≥500 years) from conversion of primary-forests have been found to be around 50 to 67% of the long-term-average primary-forest C stock (Chapter 2; Harmon and Marks, 2002). Nevertheless, differentiation of forest history is generally absent from reports of C balance from the commercial forestry sector, even though usage of wood-products from production forests is promoted as greenhouse-gas mitigation (Forestry Tasmania, 2009a; Forestry Tasmania, 2009b; FWPA, 2009; The CIE, 2015; e.g. FPC, 2016). The arguments on which such putatively positive outcomes for atmosphere conservation are based, either discount primary-forest C through generic marketing (Wilson, 2006; COAG, 2009; The CIE, 2015), discount future fire, allowing unnatural growth (e.g. MBAC, 2007), aggregate recovering- and primary-forests (e.g. MBAC, 2007), or they simply present an purportedly benign carbon-footprint as a basis for arguments of fossil-fuel substitution (e.g. Forestry Tasmania, 2010). In contrast, an independent report for the State of Tasmania shows that forest business as usual will result in net emissions (May et al., 2012a).

A major wood-product from production forests globally is pulpwood. Japan is the leading, global importer of hardwood chips (mostly from eucalypt species), these, until recently, being sourced principally from Australia (WRI, 2010), which was the largest exporter of hardwood chips in the Asia-Pacific region (Attiwill and Adams, 2008; ABARES, 2010). Australia's major exported wood-product, on both a volume and monetary basis was woodchips, with 86% of hardwood woodchips produced in 2008/09 being exported. Tasmania was the Australian State exporting the most hardwood chips, e.g. ~1.5 Tg $year^{-1}$ (dry weight) in 2009/10 (ABARES, 2010), with the majority sourced from native forests (although production from plantations had been increasing). From 1924–1939 State concessions of over 0.357 Mha of Tasmanian native forests were granted for pulpwood extraction (Gray, 1939; Helms,

1945). From the 1960s exporting dominated (Florence, 1993). The flux of C exported as hardwood chips from the whole Tasmanian forest estate (of 3 Mha) and hardwood plantations was ~0.3 Mg ha^{-1} $year^{-1}$ in 2009-10 (ABARES, 2010) (assuming 50% (w/w) of C in dry woodchips) — at least seven times greater than the flux from any other State. As a result of this activity, the production area in Tasmanian State forest (government-owned land) was 29(±6)% below carbon carrying capacity in 2009 (Appendix IV), and a vestige of that C deficit remains in wood-products (MBAC, 2007).

The pulpwood and high quality sawlogs were sourced mostly from the more-mature stands (Helms, 1945; Felmingham et al., 2004; Elliot et al., 2008) of Tasmania's wet-eucalypt forests (consisting of wet-sclerophyll-forest and mixed-forest Gilbert (1959); Ashton and Attiwill (1994)). Activity spread to the dry-eucalypt forests in the 1970s (McCormick and Cunningham, 1989). Although clearfelling of mature primary-forest is to be phased out in State forests by 2030 (Forestry Tasmania, 2009c), the atmospheric legacy of that activity to-date will be long-lived and therefore requires quantification.

The source of the larger C fluxes is important in carbon management and hence to climate change modelling. Within wet-eucalypt forests the potential C-density varies between forest communities. The trend of C-density for the three most logged dominant species is: *Eucalyptus regnans* > *E. obliqua* > *E. delegatensis* (for area-based averages) (Chapter 3). Data presented in FPA (2010) indicate that the same sequence applies to the relative area logged of those remnant community-types from 1996–2010 (22%, 9.2% and 4.6%, respectively)[6]. Thus the remnant higher-C-density wet-eucalypt forests have been logged disproportionately more than the lower ones, i.e. the wet-eucalypt forests have been high-graded. Data in FPA (2010) also

[6] In the original paper (Dean, Wardell-Johnson & Kirkpatrick, 2012, http://dx.doi.org/10.1016/ j.agrformet.2012.03.021) I had missed some data during transcription from the FPA (2010) report and mistakenly reported the ratio as 6.4%, 2.6% and 1.8%, respectively. The error was pointed out by Moroni and Neyland (of Forestry Tasmania) in an unpublished rebuttal, via the editor of the journal (personal communication). The corrected ratio written here shows the mentioned trend of high-grading even more definitively.

indicated that higher percentages of the remnant wet-forests were logged than of the remnant dry forests. Nevertheless, similar total areas of dry-eucalypt and wet-eucalypt were logged. Due to the much greater area of wet *E. obliqua* forest than of *E. regnans* forest (FPA, 2010), more carbon from the former was potentially affected by logging in that period. The silvicultural method also has an impact on C flux, with selective logging perturbing C-density less than clearfell logging, per unit area in any one logging event.

The main aim in the present study is to incorporate forest history in the quantification of the major effects of logging of primary forest on C flux. Avenues to reduce emissions and to ascertain sink capability will also be identified. The null hypotheses are: (1) that the emissions from current methods of primary-forest timber usage are recoverable in wood-products, and (2) that adjustments to forest and wood-product management cannot ameliorate emissions while maintaining supply. To focus calculations, I firstly verify whether or not the higher C perturbations were initiated from logging the wet-eucalypt forests (which have higher C-density) rather than from the dry-eucalypt forests, and verify the main silvicultural method.

4.3 Methods

Emissions are fluxes to the atmosphere integrated over time. Emissions are referred to on two temporal scales: a short-term emission (STE), being from hours to decades; and a long-term emission (LTE) dependent on pools with long half-lives (e.g. total-C, SOC, and very old wood-products), being steady-state or asymptotic over centuries or one to two millennia. LTEs are the difference between long-term, time-averaged stocks, before and after conversion to harvesting cycles. Such time-based averaging is a method recommended by the IPCC for accounting for ΔSOC: 'best estimated over several rotations or disturbance cycles' (IPCC, 2003 p 3.39), and it is equitable to also apply it to the biomass and necromass pools. Linking change in biomass with ΔSOC is recommended by the IPCC for the more-rigorous Tier-3 accounting of forest management (IPCC, 2006). The long-term, primary-forest C is a

temporal average, representative of a range of growth stages, from seedling to senescence, and including fire effects. Long-term averages of harvesting cycles are calculated at the asymptotic stage, many cycles since logging began, and include wood products. At the unit-area-level, carbon emissions can be reported as absolute values in Mg ha^{-1} or as a percentage of the original stock, and at the State-level as mass (e.g. in Tg).

The C fluxes originating from areas of specific forest-types are examined— i.e. by the IPCC 'production' approach of carbon accounting (as opposed to tallying national imports and exports) (Lim et al., 1999). The major C fluxes from commercial operations in Tasmanian native forests were analysed at two scales: (a) spatio-temporally, at the State-level; and (b) at the unit-area-level; and the two scales were then linked.

A spatio-temporal electronic database of forestry operations in Tasmania was implemented in 1999 by the Forest Practices Authority (FPA), to support (a) the inter-governmental Regional Forest Agreement of 1997 (DAFF, 2010a), and (b) the Forest Practices Code (McIntosh and Ware, 2008). That FPA database lends itself to detection of temporal trends for forest community-types and silvicultural methods. It is used to determine which commercial forestry operations were dominant on which major forest-types, then by combination with GIS-based biomass values: which of those types had potentially incurred major C fluxes. The temporal carbon trajectories for two specific forest community-types were examined at the unit-area-level. The example forests were at the high and low extremes of wet-eucalypt C-density; one being regenerated to local species and the other to plantation species— these requiring two different models. The influences examined were forest age at the time of first logging, half-lives of carbon pools, ΔSOC, harvest cycle length and wood-product pools.

4.3.1 Tasmania State-wide

A logging coupe may contain one or more forest-community types, for which the fluxes can be determined at the unit-area-level, then up-scaled. The FPA database lists for each coupe: the area, forest-community-type, silvicultural prescription, land tenure and activity start date. Records were available for 1999–2009 (11 years). All eucalypt forest communities listed as 'tall' and 'wet' in the FPA database were grouped under wet-eucalypt. Trend lines for areas of major forest-types logged (e.g. dry-, wet-, non-eucalypt), forest communities, and silvicultural prescriptions from 1999–2009, were determined by the standardised major-axis method of regression, using SMATR (Warton et al., 2006). A State-wide GIS layer, 'TASVEG' (DPIW, 2010) of the extant vegetation-types mapped at a scale of 1:25,000 between 1998 and 2009 (Harris and Kitchener, 2005) mapped forest community-types. The forest community-types are grouped into (a) wet-eucalypt forest (with minimum height at maturity of 34 m, Stone (1998)), (b) dry-eucalypt forest (less than 34 m at maturity), (c) plantation, and (d) non-eucalypt forest.

Two layers of potential biomass, derived from remote-sensing and ground-calibration, were from: (a) the National Carbon Accounting System (NCAS) (Richards and Brack, 2004) and (b) a study on anthropogenic change to vegetation (Berry and Roderick, 2006) (termed B&R hereafter). The B&R and NCAS layers were overlain on the TASVEG layer (using a GIS), and mean values of biomass assigned to wet-eucalypt and dry-eucalypt forests. The B&R layer included C in aboveground [live] biomass and roots, whereas the NCAS layer did not include root biomass. To compare estimates from the two layers, 15% (representing root biomass) was added to the NCAS layer. (The figure of 15% was an average of 10.5% for ash eucalypts (Feller, 1980) and 20% for temperate, high-biomass eucalypt forests (Mokany et al., 2006).)

Absolute values of potential biomass for wet-eucalypt in the NCAS and B&R layers have high uncertainty, and are underestimated (Richards and Brack, 2004; Berry and Roderick, 2006; Keith et al., 2010). Therefore, when calculating absolute emissions

for wet-eucalypt forests State-wide I used instead, as the long-term primary-forest average, the carbon stored in biomass determined from State-wide field-inventory for average mature (>110 years) stands from Moroni et al. (2010): 232 Mg ha^{-1}. A standard deviation for that figure, of 20 Mg ha^{-1}, was derived on an area-weighted basis from the standard deviations for wet-eucalypts categorised by canopy geometry in Moroni et al. (2010). Comparisons for the potential biomass of wet-eucalypt were: (a) the average biomass C for mature forests for southeast Australia (including oldgrowth in Tasmania) — 289 Mg ha^{-1} (Keith et al., 2010), and (b) from field inventory by Forestry Tasmania in the southern forests the biomass C for mature (i.e. >110 years), wet-eucalypt forest— 225 Mg ha^{-1} (Raison et al., 2002). Primary-forest stands of ~250 years of age can have 50% more biomass than younger 'mature' stands of ~120 years of age (e.g. Dean et al., 2004; Keeton et al., 2010). The average-aged wet-eucalypt C in biomass from Moroni et al. (2010) was 222 Mg ha^{-1}, which included substantial areas of young, commercial regeneration, and was only 4% lower than their 'mature' value of 232 Mg ha^{-1}— thus their 'mature' value may not have been representative of mature primary-forest or of natural, temporal biomass distribution— the long-term average potential biomass at the unit-area-level. Therefore, calculations of absolute emissions using that 232 Mg ha^{-1} value for the potential biomass in wet-eucalypt primary forest, were conservative.

4.3.2 Site studies— unit-area-level

Two specific, wet-eucalypt primary-forest sites in Tasmania were modelled, representing two of the major types of commercial forestry: one was converted by 'clearfell, burn and sow' (sowing of eucalypt seed) (CBS) (Ellis et al., 1982) and one to plantation (planting of germinated trees):

1) Site-1: mixed-forest, even-aged *E. regnans* forest with high rainforest understorey, senescence mid-age 500 years, senescence rate 5, understorey multiplier 0.75, Lat/Long -42º 49′ 53.07″/ 146º 34′ 17.32″, and

2) Site-2: wet-sclerophyll, uneven-aged, mixed-eucalypt species, Lat/Long -41º 42′ 55.22″/ 146º 37′ 22.18″.

The senescence and understorey parameters listed for Site-1 refer to model settings in CAR4D (Dean et al., 2004). *E. regnans* (Site-1, coupe SX004C, Figure 4-1) is one of the taller eucalypts of the ash group, common in mixed-forests and is more fire-sensitive than most eucalypts, although uneven-aged stands are presently common (Turner et al., 2009). Site-1 was converted by CBS, to harvesting at 80 year intervals.

Figure 4-1. View NNW from Site-1.
Shows a mosaic of forest cover types now typical in areas such as the Styx and Florentine Valleys in Tasmania, from which, since the 1970s, the major wood-product has been pulpwood for the export market. Mature primary-forest logging in the foreground (Site-1), plantation in the mid-ground (at 3.5 km), harvesting-cycle native forest in the upper-right mid-ground, and mature primary-forest forest in the background (at 4.2 km). The mosaic can also be viewed from above using Google Earth© with the coordinates from section 4.3.2.

The uneven-aged, wet-sclerophyll forest at Site-2 (coupe HU320a) was converted by clearfell and intense burn to an *E. nitens* plantation with a typical 15 year (Greaves et al., 2003) harvesting cycle. In modelling, the default, non-endemic species multiplier of 2.3925 was used for *E. nitens* (Waterworth et al., 2007). Site-2 had been lightly, selectively logged in ~1950. Further details on both sites, and previous modelling, are in Chapter 2.

For carbon accounting of commercial forestry, two different models were used, each suited to one of the two sites: CAR4D for Site-1 and FullCAM for Site-2. The designs and formulas for these models have been described previously: CAR4D—Dean and Roxburgh (2006), Dean et al. (2003), Dean et al. (2004); and FullCAM (version 3.13.8)—Brack et al. (2006), Waterworth and Richards (2008). The two models had different capabilities applicable to the two sites (section 1.3.6).

For Site-1 I examined model sensitivity to: variation in age at initial logging, SOC half-life, wildfire frequency in the primary forest, harvesting cycle length, and proportion of SOC lost during CBS. A cycle length of 200 years was tested (as an alternative to 80 years), as that is required for regeneration of rainforest species in mixed-forest (Hickey, 1994; Tabor et al., 2007). The increase in half-lives of wood-products necessary to maintain total-C at the [temporal-average] primary-forest level was modelled, for Site-1 and -2. As FullCAM cannot simulate some ΔSOC effects associated with logging or ΔSOC below 0.3 m, for Site-2 only biomass and wood-products were modelled, not SOC. Sensitivity of emissions to root decomposition rate for plantations was tested for Site-2.

Half-lives of wood-products can be a point of contention, and therefore, as a check on the validity of the half-life of 2 years for pulpwood from Site-1, the product was followed through the lifecycle of nominal pulp and paper mills, recursive recycling and landfill. The assumed 'in-use' half-life for the paper product was 1 year (Barson and Gifford, 1989; Markewitz, 2006) and the value of 2 years was adopted to

accommodate emissions during international freighting, milling and domestic burning.

Conversion efficiency of the first paper mill was 70% (Richards et al., 2007) and 60% for the recycling mill (e.g. PRPC (2009); risen from 40% in 1980). The C losses from conversion were assumed to be emitted within a year; thus catering for oxidation of industrial effluent and burning waste within the mill for fuel. Domestic burning of paper products for fuel, and municipal incineration in place of landfill, were included with paper-mill losses. The in-use half-life of paper was 1 year with half going to recycling and half going to landfill with a half-life of 10 years (following Markewitz, 2006). Emissions from landfill were modelled firstly as all CO_2 (no CH_4), and secondly as 75% CH_4 (25% CO_2). The global warming potential of CH_4 was assumed to be 21 times that of CO_2 (UNFCC, 2014). The effect of each CH_4 molecule on the atmosphere was simulated by converting to an equivalent loss of carbon (21- 1= 20 carbon atoms) from the total paper pool (the in-use plus in-landfill pools). Curves were fitted to the resultant two streams of total-carbon data. As the recursive recycling and two pool half-lives led to non-first-order kinetics, the following formula for decomposition was used for total carbon at time *t* (in years), C_t:

$$C_t = C_0 \exp(-kt^d) \qquad \text{Eq4-1}$$

where exp is the base of natural logarithms, C_0 the initial carbon mass, *t* is time in years, and *k* the decomposition constant, and *d* the nonlinearity parameter. Then the half-life $t_{1/2}$ is given by:

$$t_{1/2} = (-\ln(0.5)/k)^{1/d} \qquad \text{Eq4-2}$$

The result was that emission was lower (longevity higher) using the pulpwood half-life of 2 years than when using a more-detailed LCA, which gave a half-life of 1.55 years (Figure 4-2). Thus, the pulpwood default half-life of 2 years in CAR4D was appropriate and possibly conservative, and suitable for use in the present work. It is

noteworthy that, after 10 years, due to CH_4 emission, there was a net emission above that from the original C content of the wood from the forest alone.

The emissions from paper products in the State-forest report of MBAC (2007 Fig. 5) can be fitted with a non-first-order exponential of the form Eq4-1, with k=0.4329 and d=0.4543, which equates to a half-life of 2.8 years. That is longer than the 1.5 years here (with subtraction for CH_4). MBAC (2007) include CH_4 emissions from landfill but appear to not model emissions due to conversion inefficiency in paper and paper-recycling mills, and to have assumed zero incineration of waste.

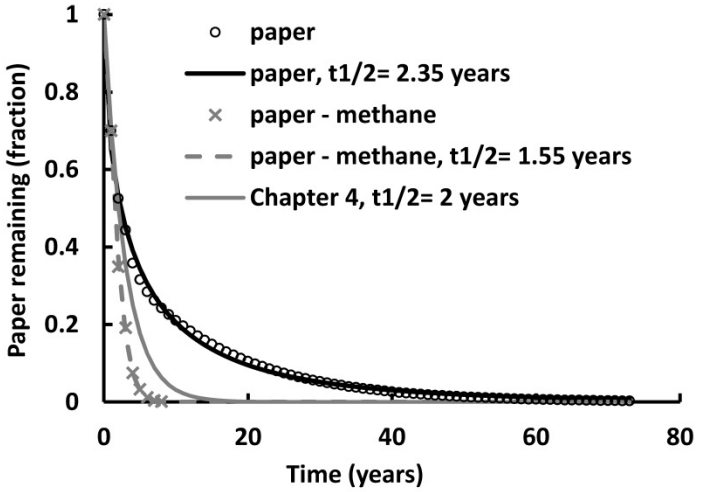


Figure 4-2. Carbon in paper (including landfill) from Site-1. Model output (markers) and fitted curves (lines). When including CH_4 (methane) emissions, and equating the CH_4 global warming potential to carbon emitted as if it where CO_2, then the remaining carbon in wood-product in-use (virgin or recycled paper) or in landfill, is significantly less. Thus the effective half-life of the wood-product is reduced.

Following the same analysis for Site-2, with FullCAM: the efficiency of the first mill was 78% instead of the 70% used in CAR4D. The $t_{1/2}$ for pulpwood used in FullCAM was 1.732, which equated to 4.72 years for landfill with no CH_4, but 2.01 years with CH_4 (i.e. the pulpwood half-life should possibly be increased by a factor

of 1.16 (= 2.01/1.732)). The default half-life was left at 1.732 years for consistency and because other half-lives were also tested. The time when CH_4 caused higher CO_2-e emissions than were in the original wood (if oxidised completely) was 9 years, i.e. approximately the same as for the 10 years with CAR4D.

The default half-life of 40 years for the sawlog product from Site-1 was taken from (Polglase et al., 1994) for *E. regnans* logged on an 80 year cycle. Their half-life was intended as an 'in-use' half-life, with a weighted mean residence time of 54.6 years, which equates to a half-life of 37.8 years. However, the sawlog product is mainly for domestic consumption and so Tasmanian mill losses, interstate freighting, and Australian product-usage losses must be incorporated into the half-life as used in CAR4D (which includes both 'in-use' and mill losses in the one half-life; section 1.3.6, section 4.3.2). Mill losses include sawdust and offcuts, which are used for fuel in mills, given to local residents for domestic fuel, and converted to chips for pulpwood and fibreboard, usually within a year of felling. These losses to lower-half-life pools are of the order of 35% to chips (or offcut equivalent) and 19% to shavings and sawdust (Jaakko Pöyry, 1999) in a sawmill with efficiency of 46%. The cradle-to-gate (forest-stand to Tasmanian chip mill or sawlog mill) diesel energy emissions are not relatively large, being only 6 tonnes of carbon per cycle (<1% of product carbon) using 354 MJ m^{-3} (May et al., 2012b) and 72.6 g-CO_2-e/Mj (BEC, 2012). All house building and old-house demolitions have waste wood, some of which goes to landfill, is burnt on-site, or burnt in domestic fires. Less is recycled or used for smaller products. Landfill emissions include the higher-GWP CH_4. Taking all these factors into account, it was considered that the grouped half-life of 40 years was appropriate for sawlog in CAR4D, including mill losses.

4.4 Results

4.4.1 Tasmania State-wide

The total coupe areas of wet-eucalypt and dry-eucalypt forests logged from 1999–2009, combined with potential biomass of these two major forest types from Moroni et al. (2010) indicated that the C in biomass subject to commercial forestry was 13% higher for wet-eucalypt than dry-eucalypt (28(3) and 25(1) Tg respectively) (Table 4-1). If it was assumed that non-clearfell logging from 1999–2009 (from thinning to aggregate retention) influenced on average only 50% of a coupe's biomass, whereas clearfell influenced 100% (apart from reserved zones) then the carbon affected in wet-eucalypt was 54% higher than for dry-eucalypt (24(2) and 16(1) Tg respectively).

Table 4-1. Tasmanian native forests logged from 1999–2009 (financial year July to June): areas clearfelled and clearfelled to plantation, and potential (mature primary-forest) C in live biomass. Affected C in biomass = area × Mel[b] carbon density. Standard deviations in brackets. ('C in Biomass, Average': average of NCAS and B&R includes an additional 15% to NCAS to account for roots. [a]: 'Affected C in Biomass equiv.' assumes 100% of clearfelled carbon and 50% of non-clearfelled carbon was affected. Mel[b]: derived from Moroni et al. (2010)).

	Area Non-clearfell (ha)	Area Clearfell (ha)	% clearfell to plantation
Wet-euc.	35204	87037	51
Dry- euc.	155904	52049	71

	C in Biomass (Mg ha^{-1})				Affected C in Biomass (Tg)	Affected C in Biomass equiv. [a] (Tg)
	NCAS	B&R	Average	Mel[b]		
Wet-euc.	90(50)	117(24)	110(41)	232(20)	28(3)	24(2)
Dry- euc.	67(39)	108(28)	92(39)	121(6)	25(1)	16(1)

Extrapolation of trends indicated that the area of wet-eucalypt logging was higher than for dry-eucalypt logging prior to 1999. It was confirmed by: (a) wet-eucalypt and dry-eucalypt operations in State forests covered 4876 ha and 2941 ha respectively in the financial year 1993/94 (L. Edwards, Forestry Tasmania, personal communication, 1995); (b) the State forest area of wet-eucalypt even-aged regeneration from 1970–1989 was 47–80% higher than for the 2000s and three to five times that of dry-eucalypt (from data in Moroni et al., 2010); and (c) 66% of the 38100 ha decrease in native forest area from 1996 to 2001 was wet-eucalypt State forest, mostly from conversion to plantation (FPA, 2010). Therefore, at least 50% more wet-eucalypt C has been subjected to commercial forestry than dry-eucalypt C, for at least two decades prior to 2010. Thus confirming, at the State-level, that higher emissions have been generated from the wet-eucalypt forests rather than from the dry-eucalypt forests, and thereby confirming that wet-eucalypt forest was the appropriate forest-type on which to focus the more in-depth calculations.

The average of NCAS and B&R carbon in potential biomass for wet-eucalypt in Tasmania (State-wide) was 110(41) Mg ha^{-1} (Table 4-1), which was less than half the value from field inventory of 232(20) Mg ha^{-1}— confirming that the absolute values in those layers were too low for wet-eucalypts, but here at the State-level.

Trends for the major forest types from 1999–2009 (Figs. 4-3,4-4, Table 4-2) were: a highly variable rate of dry-eucalypt logging; and a steady decline in wet-eucalypt and non-eucalypt logging, mirrored by a steadily increasing plantation harvest, led by *E. nitens*. The formulated trends can be used to calculate emissions for the near future, when combined with biomass values and percentage emissions.

For the majority of wet-eucalypt logged from 1999–2009 the prescription was clearfell, with a declining amount of non-clearfell (Figure 4-3.b). Clearfell also dominated from the 1960s to the 1990s (Ellis et al., 1982; McCormick and Cunningham, 1989; Hickey, 1994; Elliot et al., 2008). From 1999–2009, ~50% of wet-eucalypt clearfell was for plantation establishment, officially ceasing in State forest in 2007. Wet-eucalypt logging was less on private land, and was decreasing

Table 4-2. Equations for trends in areas logged in Tasmania from 1999–2009.

Equation forms: "1" is 'area = a×exp(b×(year-1990))', "2" is 'area = (a×(year-1990))+b'. Standard deviations in brackets. Average area logged is included for use when p>0.05.

Cover Type	Equation form	A (ha year^{-1})	B (ha)	R^2	p	Average (ha year^{-1})
Wet-eucalypt	1	33190(494)	-0.0812(0.0108)	0.84	0.000	11030(3070)
	2	-924(142)	23970(2040)	0.79	0.000	
Dry-eucalypt	2	-690(220)	28570(3190)	0.08	0.390	18910(2290)
Non-eucalypt	1	8095(404)	-0.1474(0.4487)	0.60	0.005	1141(539)
	2	-163(37)	3420(526)	0.54	0.010	
Plantation	2	644(83)	2180(1190)	0.85	0.000	11190(2140)
E. nitens Plantation	2	675(58)	-4660(837)	0.93	0.000	4794(2240)
Wet-eucalypt CBS	2	200(62)	60(130)	0.16	0.216	3473(670)
E. delegatensis CBS	2	28(8)	100(110)	0.36	0.053	502(95)
E. obliqua CBS	2	180(57)	-270(820)	0.08	0.400	2216(588)
E. regnans CBS	2	36(9)	63(130)	0.41	0.033	560(118)
Wet-eucalypt remaining cleared	2	-46(12)	1060(178)	0.36	0.053	418(153)
Wet-eucalypt to roads & quarries	2	10(3)	-86(40)	0.35	0.058	58(34)

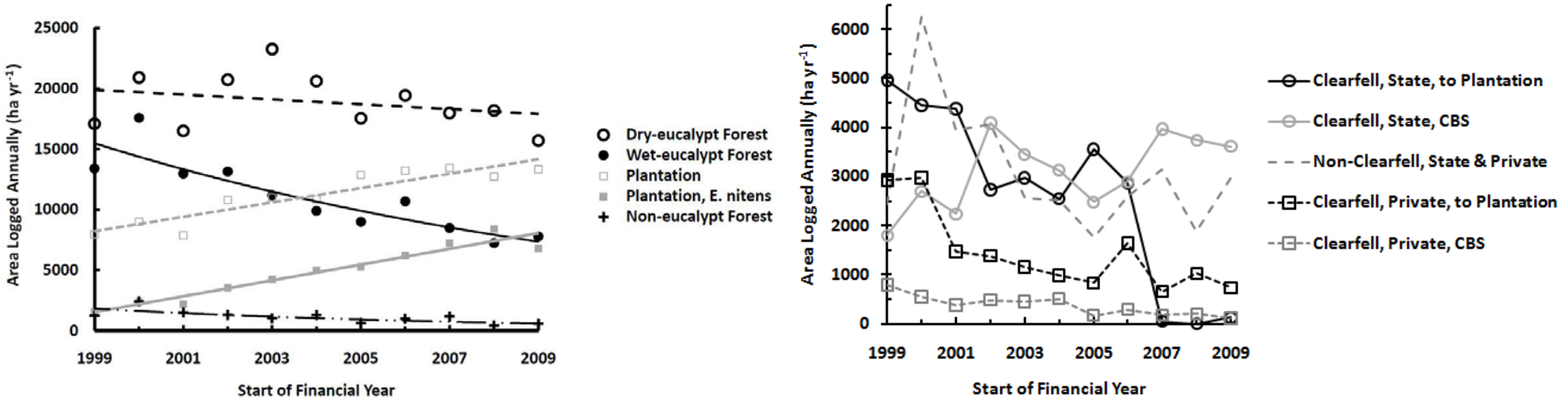


Figure 4-3. Trends in Tasmanian logging areas from 1999–2009 for major forest types, silviculture types and land tenure. Year on the abscissa is the financial year (July to June). Corresponding R^2 for lines of best fit are in Table 4-2. **(a)** All Tasmanian forests. There is an equivocal trend for dry-eucalypt forest but determinative trends for wet-eucalypt, plantation and non-eucalypt forest. **(b)** Wet-eucalypt logging. Lower areas logged on private land, and lower areas of non-clearfell. All wet-eucalypt logging decreased with time, except CBS in State forest.

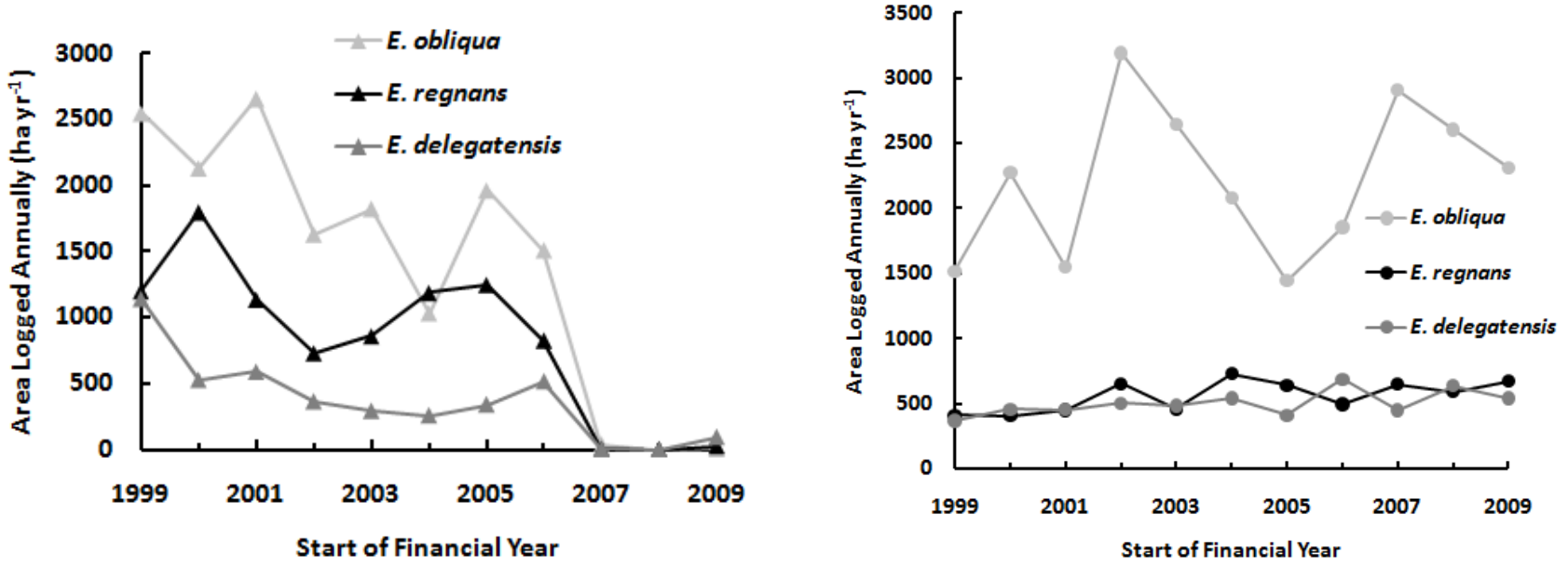


Figure 4-4. Trends in Tasmanian logging from 1999–2009 for the three most-logged wet-eucalypt community-types.

Year on the abscissa is the financial year (July to June). Corresponding R^2 and p values for lines of best fit are in Table 2. (a) Clearfelling to plantation showing relative areas and temporal decline therein. (b) Relative amounts and temporal increase for CBS.

(Figure 4-3.b). Thus the major C flux has been from clearfelling wet-eucalypt in State forest.

Although wet-eucalypt logging declined overall, there was a steady increase in wet-eucalypt CBS (Figure 4-3.b, Table 4-2), of which 90-95% was in State forest. For example, in 2009, ~640 ha year^{-1} of *E. regnans* underwent CBS logging, accelerating at 27 ha year^{-1} (Table 4-2). Approximately equal areas of wet-eucalypt were clearfelled to plantation or subject to CBS from 1999–2009, but clearfelling to plantation declined (Figure 4-3, Table 4-2). The annual area of clearfell to roads and quarries was minor compared with area logged but increased at roughly 20% per year.

4.4.2 Site studies— unit-area-level

4.4.2.1 Model sensitivities to non-wood-product parameters

Approximately ~2000 years was required to reach a hypothetical steady-state after logging began for Site-1, and 400 years for Site-2, (longer for Site-1 due to SOC.) For Site-1 the LTEs of both total-C and SOC were ~50% (Table 4-3.a). The long-term average total-C for Site-2 under harvesting, including wood-products [but not SOC], was 37 Mg ha^{-1}, compared with that for the primary forest of 127 Mg ha^{-1}, i.e. a LTE of 90 Mg ha^{-1}, or 71% (Table 4-3.b). The loss of C in biomass was high (72–86%), of which 13–15% became long-term wood-products (for Site-1 and -2 respectively).

Successive harvesting cycles did not contribute equally to the LTEs: the first cycle contributed the most, with increasingly smaller contributions from successive cycles. For Site-1 and -2 the percentages of the LTEs from the first harvesting cycle were 48% and 38% respectively, i.e. an average of 43(±5)%. Thus the LTE would be

Table 4-3. Comparison of [long-term average] C stocks and changes.

(a) Site-1. Even-aged *E. regnans*, primary mixed-forest with LUC to 80 year harvesting cycles. Half-lives: SOC 550 years, sawlog 40 years, pulpwood 2 years (including mill residues).

	Primary-forest C (long-term average) (Mg ha^{-1})	Harvesting cycle (long-term average) (Mg ha^{-1})	Δ (Mg ha^{-1})	Δ (%)
Total-C	1246	595	-651	-52%
Biomass	549	150	-399	-72%
SOC	627	326	-301	-48%
Necromass (forest debris)	67	45	-22	-33%
Wood-products	0	70	70	–

(b) Site-2. Uneven-aged, primary wet-sclerophyll forest with LUC to plantation with 15 year harvesting cycles. Total C does not include SOC. Half-lives: pulpwood 1.73 years, fibreboard 9.55 years, mill residue 0.2 years.

	Primary forest (long-term average) (Mg ha^{-1})	Harvesting cycle (long-term average) (Mg ha^{-1})	Δ (Mg ha^{-1})	Δ (%)
Total-C	127	37	-90	-71%
Biomass	121	17	-104	-86%
Necromass (forest debris)	2.4	2.2	-0.2	-9%
Wood-products	0	18	18	–

approximately halved if the primary forest was only logged once then allowed to regenerate to oldgrowth forest. Uncertainty is discussed in the next section and in the discussion.

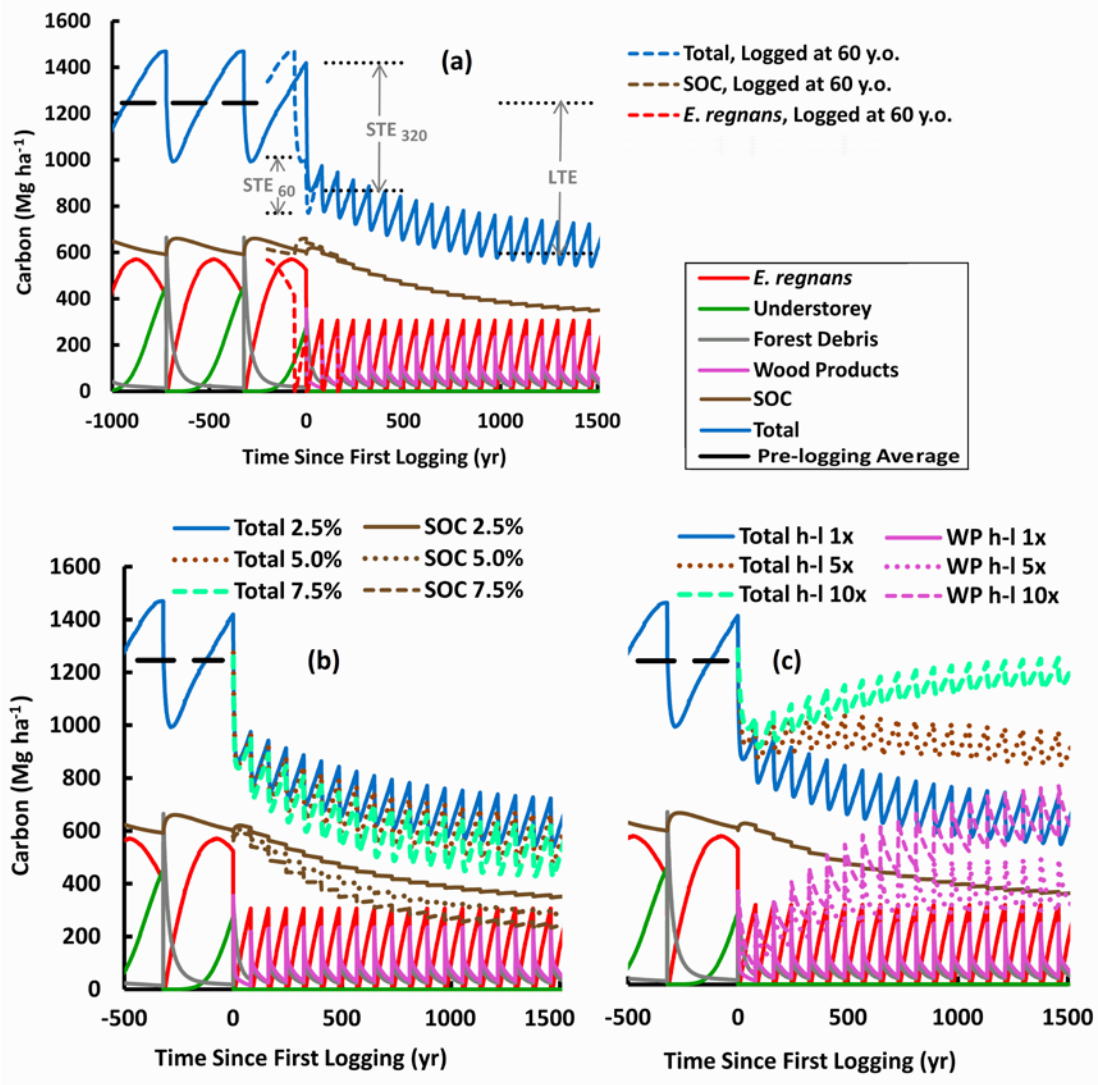


Figure 4-5. Temporal sensitivity of total-C of the harvesting cycles to variation in logging parameters for Site-1. **(a)** Logging a 60 year-old primary forest and a 320 year-old (mature) primary forest: a significant difference in the short-term emissions (STE) but equivalence in the long-term emissions (LTE). **(b)** Minor variations in SOC loss accompanying disturbance from CBS: significant long-term differences. **(c)** Variation in wood-product half-lives: recovery of the primary-forest C level (with multiplied wood-product half-lives) requires ~1,000 years.

The age of the forest when first logged (Site-1), made a significant difference to the short-term C emission: 241 and 552 Mg ha^{-1} for logging at 60 and 320 years of age respectively (Table 4-4, Figure 4-4.a). However, after two harvesting cycles the difference faded. The LTE for CBS (651 Mg ha^{-1}, or 52%, Table 4-3.a) was independent of the age of the forest at the time of initial logging: therefore the LTE refers to conversion of primary forests of any age.

Table 4-4. Comparison of short-term emissions for Site-1 (mixed-forest) from logging a 60 year-old primary forest and a 320 year-old, oldgrowth, primary forest. Total-C includes biomass, necromass, SOC and wood-products. (*cf*. Figure 4-5.a).

	Pre-logging total-C (Mg ha^{-1})	Time since logging to 1st C minimum (years)	Total-C at 1st minimum (Mg ha^{-1})	Maximum C emission in 1st cycle (Mg ha^{-1})
Logged at 60 years	1011	10	770	241
Logged at 320 years	1419	23	867	552

Variation of understorey biomass and degree of senescence for Site-1 showed that they could act in opposite directions to produce approximately the same long-term level of primary-forest C (when site productivity was constant). For example, earlier eucalypt senescence reduced the *E. regnans* C in biomass pool but that was counterbalanced by increased understorey biomass.

The LTE of ~50% SOC upon logging for Site-1 was virtually independent of the half-life of the SOC slow-pool and independent of the initial SOC stock (Figure 4-5.b). For example, the losses were 46% and 50% for SOC slow-pool half-lives of 350 and 850 years respectively. The SOC efflux magnitude and rate did however vary to a limited degree with parameters such as senescence and understorey biomass and would most likely depend on other factors such as site productivity, fire history and understorey species.

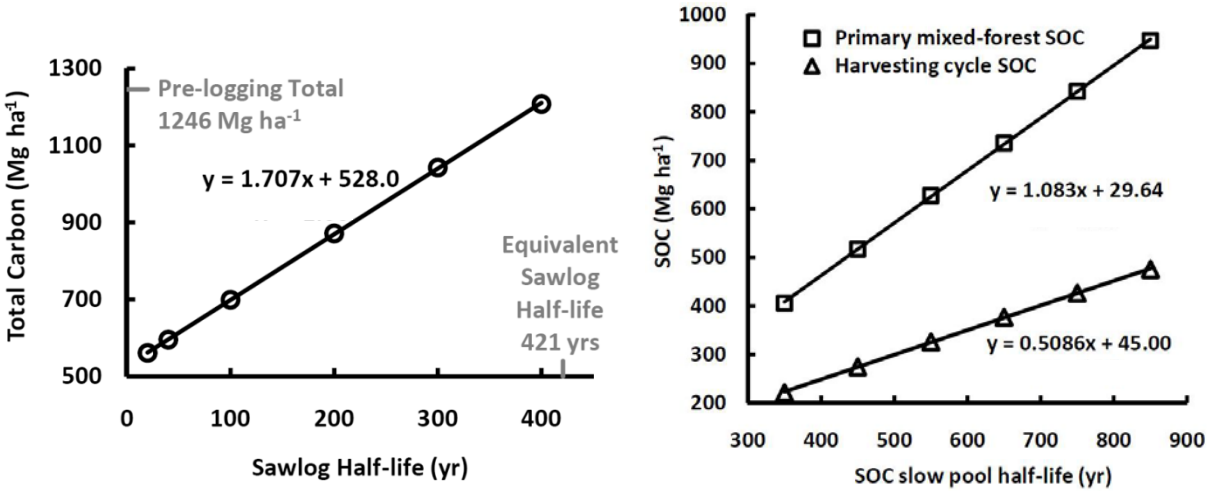


Figure 4-6. Long-term sensitivities of SOC and total-C at Site-1 to (a) sawlog half-life and (b) SOC slow-pool half-life. Note linear dependence in both cases. The default sawlog half-life must be multiplied by ~11 before the [temporal-average] primary-forest C level can be recovered. Wood-product half-life includes that of mill residue.

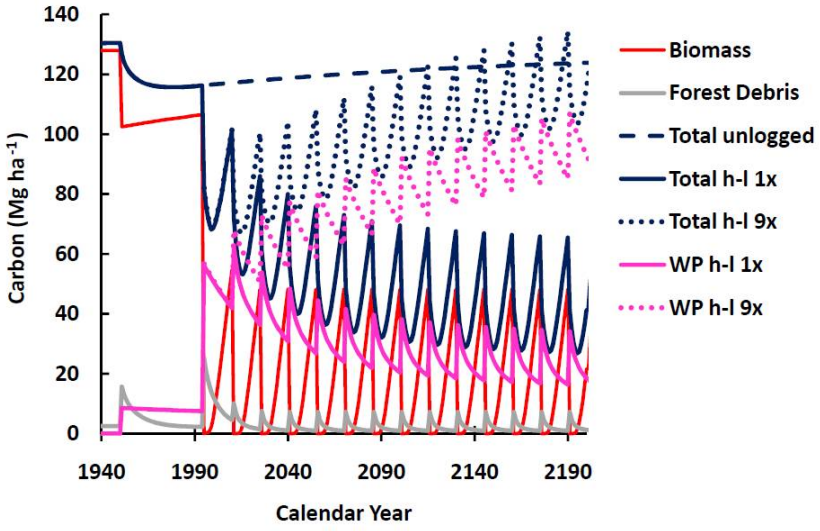


Figure 4-7. Temporal sensitivity of total-C of the harvesting cycle to variation in wood-product half-lives for Site-2. Recovery of the primary-forest C level (without the influence of SOC) requires ~9 times the default wood-product half-lives and at least ~200 years. During recovery time, the emissions would augment climate change.

Varying the fraction of SOC lost during individual CBS events had a significant impact on the long-term total-C stocks for the harvesting cycles (Table 4-5, Figure 4-5.b), namely 32% more LTE from SOC with a 7.5% loss per CBS event than with 2.5% loss per CBS event.

Table 4-5. Comparison of long-term totals for Site-1 (CBS), between harvesting-cycles and primary-forest; showing emissions, with varying disturbance influences on SOC during CBS. (*cf*. Figure 4-5.b).

SOC loss from disturbance alone (%)	Unlogged SOC (Mg ha^{-1})	Harvesting Cycle SOC (Mg ha^{-1})	Δ SOC (Mg ha^{-1})	Δ SOC (%)	Δ Total C (Mg ha^{-1})	Δ Total (%)
2.5	627	326	-301	-48	-650	-52
5	627	269	-357	-57	-708	-57
7.5	627	229	-398	-64	-748	-60

Extension of the harvesting cycle length from 80 to 200 years for Site-1 reduced the LTE from 651 to 378 Mg ha^{-1} (i.e. from 52% emission of primary-forest C to 30%), and emissions from the SOC pool from 48 to 33% (301 to 205 Mg ha^{-1}). Thus the LTE decreased by 42%; and 70% of the primary-forest C level was retained with a 200 year harvesting cycle, rather than the 48% retained with an 80 year rotation. With the longer cycle-length the long-term-average C in eucalypt wood-products dropped by 26% from 70 to 52 Mg ha^{-1}, and yielded C in rainforest species of 40 Mg ha^{-1} (error margin ~50%), possibly half of which may be merchantable.

Decreasing the duration between stand-replacing wildfires (prior to logging) for Site-1, from 500 to 200 years, reduced the long-term mean SOC pool by only 15%. That small change was because both fire frequencies allowed the forest to mature, and higher-frequency fire meant more-frequent contributions to SOC from necromass, whereas

lower-frequency fire meant higher biomass levels and hence more necromass *eventually* decomposing to SOC.

Adjustment of non-wood-product parameters for Site-2 showed that the LTE could be reduced, but only by up to ~14%, which was achieved using a 10 year half-life for plantation coarse root debris— an order of magnitude higher than FullCAM's default of 1 year. The relative appropriateness of these two alternatives is unknown and the most realistic value could lie somewhere in between.

4.4.2.2 Model sensitivities to wood-product parameters

At the unit-area-level, wood-products were equivalent to ~6.5% of the primary-forest C. They were 10% of the long-term, harvesting-cycle total-C for CBS and 25% of plantation total-C (not counting SOC), which is similar to their 16% contribution to total C in UK plantations (Dewar and Cannell, 1992).

After logging primary-forest wet-eucalypt the gradual rate of decline in total-C in the first century was reduced if more wood went into the sawlog pool than the pulpwood pool, but that effect faded after two centuries. The wood-products constituted 10% and 25% of the long-term total-C for Site-1 and -2 respectively (when SOC was included). That stock was only ~6% and 7% (respectively) of the primary-forest C level.

There was no significant difference to LTE for Site-1 when default wood-product half-lives were doubled, even without mill losses (Figure 4-8).

The primary-forest C level could only be recovered after conversion to harvesting cycles if unrealistically-high wood-product half-lives were deployed, namely ~10 times the default, for both CAR4D and FullCAM (Table 4-6, Figure 4-4.c, Figure 4-5.a, Figure 4-7). For example, the sawlog half-life for Site-1 would have to be ~421 years to

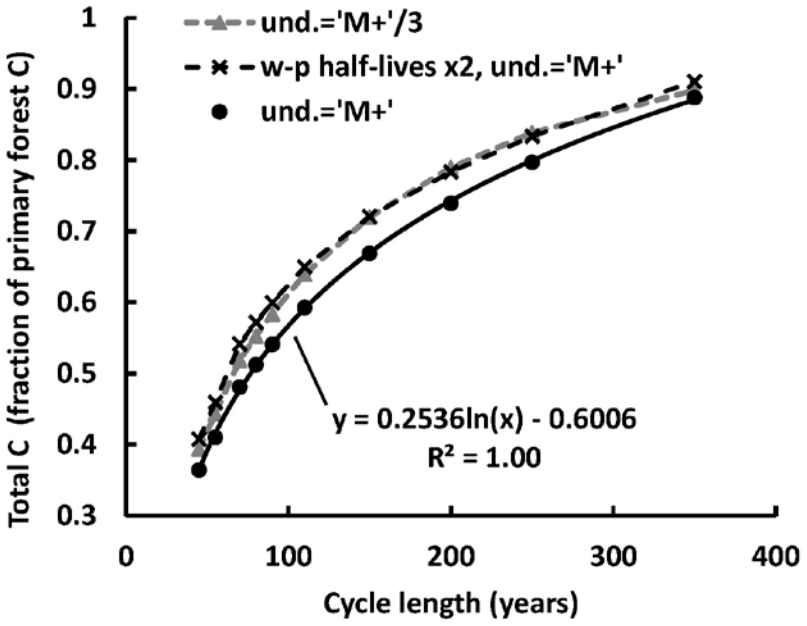


Figure 4-8. Long-term total C (fraction of primary forest long-term C) versus harvesting cycle length. The black, bottom line is for Site-1; the grey line is the same but with 1/3 of the rainforest understorey biomass (such that it cannot alone recover stand-level biomass immediately after eucalypt senescence, e.g. as for a Florentine coupe (Chapter 3); and the black, top, dashed line is for Site-1 but with wood-product half-lives doubled (i.e. 80 years for sawlog and 4 years for pulpwood, including mill residues). This shows: (a) a way to re-sequester primary forest carbon is by extending both wood-product half-lives and harvesting cycle length; (b) the effect of doubling wood-product half-lives on re-sequestering primary forest C is merely similar to the effect of extending the harvest cycle length from 80 to 110 years; and (c) the measurable effect of doubling of wood-product half-lives is similar to the variability in contribution to total C from rainforest understorey biomass.

recover total-C to primary-forest levels. Similarly for Site-2: half-lives of over 16, 86 and 1 years for paper, fibreboard and mill residue respectively (but mill residue was burnt at the mill) were required to recover primary-forest C levels. These results indicate that emissions cannot be recovered in wood-products.

Table 4-6. Influence of altered half-life of wood-products.

(a) Site-1. The CAR4D defaults of 40 years and 2 years (for sawlog and pulpwood respectively) were multiplied by the same factor (column 1). The long-term total-C after logging can be compared against the unlogged, long-term total-C of 1218 Mg ha^{-1}. (*cf.* Figure 4-4.c, Figure 4-5.a).

Multiplier	Sawlog Half-life (years)	Pulpwood Half-life (years)	Long-term C in Wood-products (Mg ha^{-1})	Long-term Total-C (Mg ha^{-1})
0.5	20	1	35.8	561
1	40	2	70.3	595
2.5	100	5	174	699
5	200	10	346	871
7.5	300	15	517	1043
10	400	20	683	1208

(b) Site-2. Only when the default half-lives of wood-products were multiplied by 9 did the total-C recover to the unlogged level. Values for the plantation are harvest-cycle averages. (SOC not modelled.)

		Total-C mature primary forest (Mg ha^{-1})	Total-C under harvesting-cycles			
			With default half-lives		With default half-lives x9	
			Mg ha^{-1}	Δ from unlogged (%)	Mg ha^{-1}	Δ from unlogged (%)
Year	2200	124	40.1	-68	115	-7.1
	2400	127	36.8	-71	124	-1.7

With such long half-lives, the majority of the total-C was in wood-products, namely 50% and 83% for Site-1 and -2 respectively (although ignoring SOC for Site-2). The wood-product with the higher half-life dominated the wood-product stock. Half-lives

greater than ~10 x the defaults allowed more C to be stored under harvesting cycles than in the original primary forest.

After the initial logging, an exponential curve could be fitted to the total-C stock of the subsequent harvest cycles: $total\text{-}C = a + b(1\text{-}\exp(\text{-}kt))$ where a and b are constants related to the stocks at the first and asymptotic harvest cycles (with b positive for increasing stocks and negative for decreasing stocks), k is the rate constant, and t is the time since first logging. The half-time for total-C to reach an asymptote, $t_{1/2}$, was $-\ln(0.5)/k$.

For example, by least-squares regression on CAR4D output, for Site-1: a= 905.1 Mg ha^{-1}, b= -323.4 Mg ha^{-1} and k= 0.001604 year^{-1}: giving $t_{1/2}$ = 432 years, which is similar to that of the slowest emitting C pool— SOC — with a half-live of 450 years. When the default wood-product half-lives were multiplied by 10 (for recovery of total-C), then a= 905.1 Mg ha^{-1}, b= 303.7 Mg ha^{-1} and k= 0.001748 year^{-1}: giving $t_{1/2}$ = 397 years, which is similar to that of the slowest accruing C pool— sawlog — with a half-live of 400 years.

When the total-C was recovering to primary-forest levels during successive harvesting cycles, through unrealistically high half-lives for wood-products (Figure 4-4.c, Figure 4-5.b, Figure 4-7), then the recovery time was correlated to the half-life of the longest-lived wood-product. For example, when the wood-products from the plantation (Site-2) were pulpwood and fibreboard, with half-lives of 16 and 86 years respectively then $t_{1/2}$ (for recovery of total-C) was ~80 years, but when construction timber (half-life of 309 years) replaced fibreboard then $t_{1/2}$ (for recovery of total-C) was ~400 years. This meant that the time taken for the total-C stock to recover to the original, primary-forest level was similar to the duration for decomposition of the wood-product. For example, that duration was ~1,000 years for sawlog and ~200 years for the pulpwood plus fibreboard. Therefore, even if the total-C could be recovered to primary-forest level through wood-products [with unrealistically high half-lives] then recovery would take several

centuries. Such relationships emerge from the mathematics of single exponential, first-order decomposition and accumulation (e.g. Olson, 1963; Dewar, 1991).

4.4.3 State-wide fluxes

Combining the wet-eucalypt mature biomass C of 232 Mg ha^{-1} (Moroni et al., 2010) with the wet-eucalypt SOC of 271 Mg ha^{-1} (Chapter 2) (i.e. total-C of 503 Mg ha^{-1}) and the clearfell area from 1999–2009 of 87037 ha (Table 4-1), gives 43.8 Tg of C in wet-eucalypt forest affected by clearfell in that period.

With the finding of a LTE of 50% (conservatively) of total stocks for conversion to harvesting cycles (52% for CBS and 71% for conversion to plantation), then the absolute LTE from wet-eucalypt conversion by clearfell initiated from 1999–2009 will be 21.9 Tg, averaging a LTE of 2.0(±1.6) Tg from each year's operations. The approximations used, which are necessary in the absence of more definitive data, add uncertainty to the estimates and result in the large error margins assigned: 80%. The value of SOC in wet-eucalypt forests (Chapter 2) was from data for only the top 0.3 m, whereas CAR4D models SOC over the full soil profile— thus there would be an additional long-term SOC emission, from below 0.3 m.

In State forest there is 0.147(±0.0015) Mha of even-aged wet-eucalypt forest (~26% of the wet-eucalypt in Tasmanian State forest) of known age, converted prior to 2010 (Moroni et al., 2010), most probably by CBS. Following the same calculations as for 1999–2009 above, the LTE from that earlier conversion will be 37(±30) Tg and the emission from the first cycle of conversion was 16(±13) Tg. Therefore, if that 0.147 Mha is allowed to return to forest with natural regeneration cycles then a LTE emission from further harvesting cycles of 21 Tg, can be avoided (37 minus 16 Tg). Alternatively, if it is placed on 200 year cycles rather than 80 year cycles (30% emission rather than 50%) then an LTE of 15(±12) Tg can be avoided.

4.5 Discussion

4.5.1 Impacts of major fluxes and their uncertainty, on carbon accounting

Generally, error margins and uncertainties are high in forest carbon accounting. Sources of error for particular C pools, and model aptness, have been described by numerous authors (e.g. Dean et al., 2004; Köhl et al., 2008; Lindner et al., 2008; Nabuurs et al., 2008; Seidl et al., 2008) and in Chapter 2. Climate change itself produces another area of uncertainty, because increasingly, measurement of C flux and its forecast under forest management will have to compensate for the increasing background fluxes from climate change, e.g. the reversal of a growth trend across temperate and boreal Eurasia (Piao et al., 2011a).

A percentage change was used rather than absolute values to reduce the impact of uncertainty in initial stocks. Nevertheless, small uncertainties in fractional emissions corresponded to ~30% variability in long-term ΔSOC, for example— a significant level of uncertainty in the longer-term. Errors in decadal absolute emissions at the State-level, were as high as 80%. Climate change effects were not included here in the calculation of LTEs as the relative impact on logged and unlogged forests was not determined. The fact that two distinct models for two different forest types, both yielded the same result with regards to the changes in wood-product half-lives (including waste material) necessary to re-sequester the long-term emission, reinforces the significance of the finding. Error margins in observations that went into the underlying model construction (such as on the parameters for stand density as a function of age in CAR4D) were not carried through to the final results provided here—that would've been a major programming exercise beyond the scope of the present work, similarly for an analytical

solution rather than the deployment of annual step modelling. Additionally in biological systems there are correlations between effects that would have to be taken into account if propagating errors within the model. The discourse on the variability of half-lives depending on mill processes shows that some error margins would've have necessitated vague guesses and would have therefore been inappropriate to include amongst other error margins within the model. Instead the sensitivity analyses were provided to indicate the variability in the results for variation in parameter settings.

The LTE (conservatively 50% of primary-forest C) for even-aged wet-eucalypt was independent of the forest age at which logging occurred, although STEs are approximately twice as high from older primary forests. That independence arose from the use of [long-term] temporal-average stocks. An LTE of 50% is equivalent to that measured for land-use change such as to agriculture (Bolin and Sukumar, 2000). Conservation of primary forests, for their C stocks, regardless of forest age, has been suggested as a carbon conservation mechanism by Lovelock (2007) and Woodwell et al. (2011). The finding of a stand-age-independent LTE upon conversion to harvesting cycles supports that principle.

The first cycle of logging oldgrowth contributes 43(±5)% to the LTE. The initial emissions upon CBS arise from the intense burn of the logging slash (also known as 'forest residue'). In wet-eucalypt the C in slash has been measured at 255–387 Mg ha^{-1}, or 30–70% of aboveground biomass, with 58–63% of that emitted in the burn (Slijepcevic, 2001; Green, 2002; Ximenes et al., 2008a), i.e. ~at least one quarter of C in aboveground biomass is emitted within two years. Slash necromass is a function of pre-logging biomass (Ellis et al., 1982), the proportion of timber recovered, the relative biomass of non-target species, and stand age (Ximenes et al., 2008a). A wet-eucalypt forest logged at peak biomass will have the highest STE. For Site-1 that peak was at 243 years of age, but *E. regnans* can live for 600 years (Wood et al., 2010) and a mixed-forest with senescing eucalypts may have a later peak upon maturity of the rainforest

understorey after 400 years (Dean et al., 2004). Therefore a large range of forest ages may have high STEs.

The modelled SOC efflux [for the full profile] of 2.5% due to fire and disturbance with each CBS event, appears conservative (e.g. compared with 20% loss to 0.1 m, Tomkins et al. (1991), Chapter 7). The resultant LTE of 50% for ΔSOC (from repeated losses over numerous cycles), appears plausible, for example compared with -15% to 0.2 m after four cycles with only 40% biomass removal, no burn and low disturbance (Johnson et al., 2010). Processes within the model CAR4D have been checked by applying a new model, 'ForestHWP' developed by CSIRO, and, using the same half-lives: the results for SOC decline under repeated harvesting cycles were replicated (Roxburgh and Crawford, 2016).

Within ~10 years after the first logging, SOC has a post-logging, recovery peak (Figure 4-5 and Johnson et al. (2010)) before a longer-term decline (e.g. Gough et al., 2008; Xiang et al., 2009). Difficulties in measurement of ΔSOC <10% with each logging event (Johnson et al., 2010), and observation of the short-term fluctuations, could explain reports of null and minimal change (as in Guo and Gifford, 2002) (see also Chapter 7). CBS involves mixing of logging debris and coarse charcoal into the soil (Hopmans et al., 2005; Knicker et al., 2005). However, in CAR4D, carbon in slash is attributed to necromass pools rather than SOC, which could cause some discrepancy between ΔSOC measurement and modelling.

The SOC LTE from CBS is caused by disturbance and fire during each CBS event, and by the reduced returns to SOC from the time-averaged lower biomass over the harvesting cycle, compared with that in primary forest. The heat from CBS can dissociate mineral carbonates (with CO_2 efflux), and induce post-fire erosion of clay (with subsequent reduced organic retention capacity) (Tomkins et al., 1991; McIntosh et al., 2005; Terefe et al., 2008). However, in CAR4D the nature of the soil is considered

time-invariant. Thus, the intense silvicultural fires may decrease SOC in the long-term more than the modelling indicates.

LTE and STE were modelled using conservative assumptions for all the uncertain variables, which provides validity to the major conclusions in this study.

ΔSOC efflux during CBS requires targeted experiments to determine magnitudes of the contributory affects, especially as minor variations give significantly different LTEs of SOC.

4.5.2 Silviculture and primary-forest C levels

The substantial primary-forest component (a legacy or core) of C that is retained after the first logging is depleted in subsequent harvests, producing successively smaller emissions (Figures 4-5, 4-6). That core allows equivalence for LTEs whether the primary forest is logged prior to maturity (e.g. at 60 years of age), at peak biomass, or when senescent. The landscape-level mosaic of primary forest, CBS-forest and plantation (Figure 4-1), combined with the attrition of primary-forest C, means that the landscape-level C stock (including wood-products) is below that prior to commercial forestry and will decline further.

Under CBS, production of rainforest species and wet-eucalypt biodiversity can be sustained with harvest cycles ≥200 years (Hickey, 1994; Brown, 1996; Tabor et al., 2007). Uneven-aged stands generally have higher biomass than even-aged stands, peaking when older individuals are 'silviculturally over-mature' (Knuchel, 1953; Kurz et al., 1996; Phillips et al., 2002). In terms of biomass the rainforest species in an even-aged mixed-forest are a counterpart to the younger eucalypts present in an uneven-aged forest. The 200 year frequency coincides with decreased LTEs, by 30–50%; for example avoiding C efflux of ~15 Tg for ~26% of wet-eucalypt in State forest. The finding of

higher total-C (forest *in situ* plus wood-products) with longer harvest cycles is consistent with other findings (e.g. Liski et al., 2001; Dean and Roxburgh, 2006). However, when mature-tree biomass is not represented or post-harvest burns are omitted (for example), then the benefit of longer cycles becomes indeterminate (e.g. as in Spring et al., 2005; Ranatunga et al., 2008). Although the long-term eucalypt wood-product levels are 26% lower with 200 year cycles, it would allow production of the rainforest 'specialty timbers' (~40(±20) Mg ha^{-1}) and the prized, large-dimension eucalypt timbers. Although both cycle lengths have almost identical STEs, some stands currently nearing the end of their first harvest cycle would have logging delayed [under a 200 year cycle], and therefore near-future emissions at the landscape-level could be reduced.

There are myriad additional influences on the LTEs not calculated here, that will affect ground truthing of modelling. Mixed-species forests can have higher SOC stability (Jandl et al., 2007). Rainforest elements protect C in the wider landscape from fire (Lindenmayer et al., 2011). CBS can create poor regeneration of the rainforest species and create stands more favourable to pulpwood than sawlog production (Taplin et al., 1991; Jordan et al., 1992; Tabor et al., 2007). As the seed source for the rainforest understory becomes restricted spatially with fragmentation of mixed-forest (Tabor et al., 2007) it will become more-difficult to regain primary-forest C levels. Maintaining biodiversity and reducing fragmentation of forests helps to reduce the vulnerability of ecosystems to climate change and wildfire (Lindenmayer et al., 2009; Pittock, 2009; Lindenmayer et al., 2011). In Chapter 2 it was shown that climate change will reduce SOC in the wet-eucalypt forests in the long-term. The process there is similar to the forecast the attrition of rainforest by fire from climate change (Riddington, 2014), which often neighbours mixed-forest in Tasmania.

4.5.3 Wood-product half-life and emission recovery

Emissions from primary forests are not recovered in wood-products under current management and recovery requires a ten-fold increase in wood-product longevity. Due to the mathematical equality between wood-product half-life and the time to maximum sequestration of C in wood-products, even if the half-life could be increased ten-fold, the primary-forest C level would still not be reached for several centuries (~1,000 years for sawlog and ~200 years for pulpwood plus fibreboard). That time is necessary to transfer C from biomass, with substantial losses along the way, to fill one new pool, namely wood-products. During recovery, emissions would contribute to global climate change positive feedback, inducing further emissions globally. (Positive feedback on anthropogenic emissions will in effect increase them by 18(±11)% by 2100 (Denman et al., 2007)). The offsetting of this feedback would require wood-product half-life to increase even further.

The high wood-product half-lives found necessary to recover primary-forest C levels—365, 86, 16 years for sawlog, fibreboard, and paper respectively, are unrepresentative of contemporary values. For example, a sawlog half-life of 365 years could only be achieved if all sawlog from the coupe gate incurred no mill residue or other downstream losses and decomposed no faster than that reported for the Sydney landfill sites in Richards et al. (2007). For Tasmanian wet-eucalypt logging, because of the high export pulpwood component, accounting must include the half-lives of overseas mill residues and products, cargo ship emissions and anaerobic decomposition of wood-products in overseas landfills. The methane of such anaerobic decomposition may increase the climate change impact of wood-products and counterbalance their below-ground longevity (e.g. Pingoud et al., 2001). In the check of CAR4D by using the new model 'ForestHWP' with the same half-lives (Roxburgh and Crawford, 2016) the results for the need to radically extend wood-product half-lives to recover primary-forest C were replicated.

Three options are considered for increasing wood-product half-life. Reduce slash by mechanisms such as reducing 'smash' upon felling and reclassify more of the felled timber normally added to slash and burnt. Reduce mill residue by mechanisms such as salvaging off-cuts normally burnt or used as pulpwood. Recycle existing wood-products.

If more slash is converted to wood-product, the net C balance may not improve, however, because: (a) the long-term SOC stocks on-site will be lower (e.g. Raison et al., 2002); and (b) the some slash may have a longer half-life than that of pulpwood (e.g. Dewar, 1991).

Wood-product recycling extends the half-life directly and reduces demand for fresh product. Recycling was nominated as beneficial for most secondary forests internationally, in preference to burning waste paper or moving it to landfill (Villaneuva and Wenzel, 2007). Housing demolitions accompanying urban consolidation are commonplace in Australia, with wood mass in landfill from construction and demolition totalling ~1.8 Tg $year^{-1}$ (Taylor and Warnken, 2008). Where the driver of logging is sawlog quota then recycling a proportion of construction timber should reduce the annual area logged by an equivalent proportion. But if recycled wood-products are used to increase supply, rather than to substitute for it, then emissions will not be reduced.

Emissions can also be lowered by substituting such wood-products with lower-C-footprint or longer-lasting materials, such as locally-sourced stone; E-crete, which has 20% of the emissions of Portland cement (van Denter et al., 2010); recycled concrete; plantation composite-products (grown on old-field [agricultural] land (e.g. Paul et al., 2006)); urban-grown timber (French, 1980; Brack, 2002); and straw pulp from post-harvest cereal-crop waste (substituting for pulpwood) (Kissinger et al., 2007).

In paper mills, ventures to increase wood particle recovery, decrease toxic effluent, and increase in fossil-fuel substitution, have often led to either equivalent or increased GHG emissions (e.g. Menzes et al., 2011). Notably, Mathieu et al. (2012) found that it was

better to burn waste paper than to place it in landfill, though that burning was used to substitute for fossil fuels. Substitution using cleaner energy was not assessed.

A particular version of product substitution is present in some literature on the carbon-footprint of forest industries (e.g. Naqvi et al., 2010; Lippke et al., 2011), where the 'substitution effect' pertains only to when wood products putatively produce less carbon than alternative building construction products, namely standard concrete, steel and aluminium. It may well be that wood products from plantations or previously logged forest are less carbon polluting than some of these alternatives (when SOC and biomass from the previous forest cover are discounted, or fertiliser is added), but this is not the only question to be addressed. Significantly, it remains to be proven that metal products or concrete cause more atmospheric change than converting primary forest to a logging cycle.

The use of the C-footprint from the plantation industry on long-cleared land (i.e. discounting past land cover) is inappropriate for primary-forest-logging (e.g. Harmon et al., 1990; Sathre and O'Connor, 2010; Chapter 4; Lindenmayer and Laurance, 2012) (Figure 4-9). Similarly, life-cycle-analyses (LCA) that claim emission offsets for burning wood waste [from logging, i.e. 'forest residue' or 'mill residue'] for energy actually require accounting of the C balance of logging, and comparison with other energy sources such as hydroelectricity, wind and solar power. Comparable system boundaries in the LCA must be used for any comparison with substitutable materials (Horne et al., 2009). The emissions in processing and delivering the wood-product must be considered (Ingerson, 2011), just as they are in determining the C-footprint of other industrial products. LCA will include diesel usage in the coupe, in haulage (e.g. May et al., 2012b) and in international freighting, virgin- and recycled-product mill energy usage, and NOx emissions. Such additional emissions are high for paper production, the major endpoint of Tasmanian primary-forest conversion, and for recycling. They are particularly high in the current situation in which wood-product movement occurs from

Tasmania to Japan, then to China and other countries, and then back to China for recycling (Stafford, 2007).

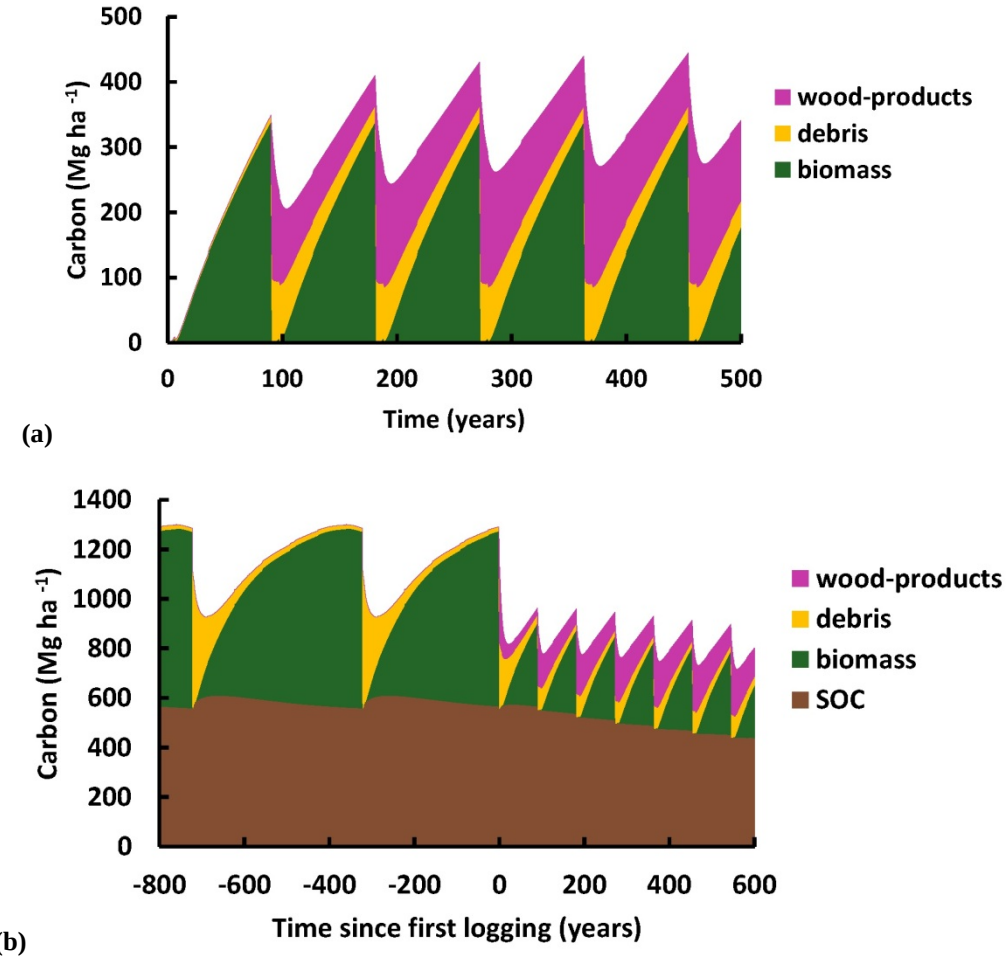


Figure 4-9. Output from CAR4D for Site-1, drawn in 'stacked-area' format: **(a)** ignoring the primary forest carbon (i.e. zero baseline), and **(b)** not ignoring primary forest carbon. Style (a) is seen in industry advertisements (e.g. Ximenes et al., 2008b Figure 10) and emphasises the temporarily increasing wood-product pool— giving the illusion of increases *ad infinitum*. It would be appropriate to use style (a) for plantations on long-cleared land, but not for LUC of primary forests. Parameters were generously set to a cycle length of 90 years and with wood-product half-lives doubled to 80 years for sawlog and 4 years for pulpwood (including mill residues). For recovery of primary forest carbon stocks with this cycle length, wood-product half-lives would still have to be somehow increased to 400 and 20 years for sawlog and

pulpwood respectively (including mill residues) and it would take ~1260 years for recovery to primary-forest levels (meanwhile emission would be contributing to climate change); international freight emissions would need to be ignored, and NOx emissions ignored.

The effects of NOx emissions may be particularly important. The C:N ratio for forest biomass is typically taken as 100 (e.g. Kreileman and Bouwman, 1994) but estimates of the global warming potential of nitrous oxide range from 153 to 310 (IPCC, 2007). Thus, the impact on climate change of NO_x emissions from logging may be at least the same order of that of CO_2. Calculations for secondary-forest timber harvesting in the USA showed approximately equal life-cycle emissions of NOx and CO_2 (Johnson et al., 2005). This would scale-up the net emissions calculated previously in the present work.

It is unlikely that adjustments to forest and wood-product management can ameliorate emissions while maintaining supply, unless those adjustments are major.

Envisaging partial change, Australian governments funded research into 'variable retention' logging (DAFF, 2005; McLarin, 2008). It was not fully implemented however because clearfelling was still required to meet the annual sawlog quota, and to provide the more-merchantable oldgrowth (Felmingham et al., 2004; McLarin, 2008); juvenile eucalypt growth rate was lower (Neyland et al., 2009); and extra road making and more careful management of fire would be required in the more-fragmented landscape (McIntosh and Ware, 2008; Neyland et al., 2009; Baker and Read, 2011). The substantial investments in the Tasmanian forest industry made by governments (Gray, 1939), financial institutions (Florence, 1993; Clark, 2004), and business (Clark, 2004), require rapid return, best provided by CBS (Clark, 2004) and impede change. Societal adaptation is necessary to recognise the 'common property' of the forest (Roberts, 2004) and atmosphere, and to provide cornerstone science for improved management (French, 1980; Durant et al., 2011). The major adjustment necessary could be stimulated by pricing forestry emissions and valuing retained C.

4.5.4 Further research

Uncertainty in primary-forest C stocks is echoed in emission calculations, especially for absolute values, and better accounting for forestry-induced C emissions is required (Anderson et al., 2010; Law and Harmon, 2011). Examples of uncertainty in wet-eucalypts biomass are: 'noneucalypt...standing-tree volumes are derived from expert opinion' and 'little to no data was available for ... non-merchantable biomass components of eucalypts', Moroni et al. (2010, p6-7)). The 'non-eucalypt' (i.e. understorey rainforest species in mixed-forest) has been measured at up to 50% of the total biomass prior to onset of substantial eucalypt senescence (Green, 2002) but the proportion could be higher, as by definition, the eucalypt canopy-cover in mixed-forest can be as low as 5%. Nevertheless, species-specific allometrics are unavailable for many of the understorey species, including that with highest biomass— myrtle (personal communication, Forestry Tasmania, 2009).

Quantitative accuracy for SOC fluxes with forestry operations is low (Peltoniemi et al., 2004; Diochon et al., 2009). Owing to the significant, long-term impact of small variations in ΔSOC accompanying CBS, empirical findings of silvicultural ΔSOC must be refined, with separate experiments for the various influences (mechanical disturbance, pre-fire drying, slash-soil mixing, fire) and input from self-thinning and root turnover in the regenerating stand; both for the STEs (as in Xiang et al. (2009)) and for the LTEs. Transects spanning the coupe boundary and an escaped burn [into primary forest] would provide site pairing (Chapter 11). Measurement of SOC to at least root-depth under mature trees is required (Chapter 8). Other items requiring further study are: (a) half-lives of coarse root debris and exported wood-products (including mill efficiencies, landfill half-life, durations when paper is in storage, landfill fraction emitted as CH_4, international transport emissions, portion historically burnt for domestic

fuel), (b) simulated burning, and purposeful slash distribution. This would help to (for example) reduce the variability around the half-lives used in modelling.

For at least the last two decades, emissions have been higher from wet-eucalypt forests than from dry-forests. Clearfelling, mostly on public land, has initiated a substantial C emission process, which has not been recovered in wood-products, and which is unlikely to be recovered without substantial change in management. Attention to quantifying that process is required.

Extending the half-lives of wood-products via recycling can reduce emissions, if used to redirect fresh timber to longer-half-life products or to substitute for future logging, rather than to increase supply. Product substitution studies must consider lower-C-footprint alternatives to wood-products. Extension of the harvesting cycle length for mixed-forests to 200 years can reduce LTEs significantly. Although reducing eucalypt wood-product stocks by ~25%, it may help sustain the ecosystem and production of rainforest-species timbers. Recognition of primary-forest C also requires recognition of the remnant core C in forests that have been logged only once.

Without improved and coordinated management, primary-forest C will undergo further attrition, contributing further to global warming. Any comprehensive global accounting of greenhouse gas fluxes requires further understanding of the effects of the interaction of logging cycles with primary forest.

Chapter 5 The effect of large trees in primary forests on soil carbon stocks and land-use emissions. Part 1: individual tree 3D modelling

5.1 Abstract

Particularly strong uncertainties for the contribution of LUC to the anthropogenic component of climate change pertain to subterranean carbon loss resulting from conversion of primary forest to secondary forest. The trunks of large trees in such primary forests are foci for the genesis of biogenic carbon. Soil organic carbon (SOC) and coarse roots are more concentrated closer to the trunks, but during soil carbon sampling the area close to and under large tree trunks has been systematically avoided due to physical sampling difficulties. Soil carbon estimates have not been correspondingly adjusted. This chapter presents a methodology to acquire high spatial resolution, 3D models of the buttress and roots of large trees, and shows how those data can be used to develop allometric relationships linking typical forest tree measurements to soil organic carbon measurements. Using terrestrial digital-single lens reflex photography and photogrammetry software (Photoscan), 3D models of tree buttress regions and roots of large trees in primary forest were made, from 29 in situ large eucalypts (including 25 *Eucalyptus regnans*), 10 in situ understorey trees, eucalypt roots, loose logging debris, fallen trees, branches, root and trunk slices, and soil profiles. Formulae were developed to link tree architecture, adjustments to SOC measurements, and easily measured forest data— diameter at breast height and ground slope. SOC was measured in the aboveground humus mound of *E. regnans* trees. The formulae linking the geometry and SOC are applicable to stand-level carbon accounting.

5.2 Introduction

The high-ends of allometric formulae, which are represented by large trees, are needed in order to establish the carbon dynamics of primary forests. However, the larger trees typical of primary forests have become less common with LUC (land use change, as defined in Chapter 1) and climate change (Galbraith, 1939; Rankin, 1947; Ferguson, 1948; MacFie, 1999; Laurance et al., 2000; Herrmann, 2006; Lindenmayer et al., 2012; Hansen et al., 2013; Jacob et al., 2013; Balmer et al., 2015; Mackey et al., 2015; McIntyre et al., 2015). A prerequisite for determining the stand-level effects of large trees on soil organic carbon (SOC), and thereby improving LUC calculations, is characterisation of the macroscopic tree-soil interface. The lower trunks of large trees have only recently begun to be measured near the ground in a way that can assist carbon dynamics calculations (e.g. Dean, 2003; Chave et al., 2005; Dean and Roxburgh, 2006; Nogueira et al., 2006; Ngomanda et al., 2012; Sillett et al., 2015). Moreover, the root volume near large tree trunks, which displaces SOC, has rarely been measured.

This chapter examines tall-eucalypt forests with a rainforest understory (mixed-forest) in the maritime-temperate climate of Tasmania, Australia. The focus is on *Eucalyptus regnans*-dominated mixed-forest. The diameter at breast height (DBH, at 1.3 m) of *Eucalyptus regnans* (swamp gum/mountain ash) may reach ~7 metres in Tasmania, and in Victoria the historical maximum DBH was 10.8 m (Ashton, 1975a). However, the older stands in Victoria containing such trees have been replaced by livestock farms (Ashton, 1975a; Beilin, 2007; Mainville, 2007). Logging of primary *E. regnans*-dominated forests has been extensive in the States to which they are native, Victoria and Tasmania (Chapter 1; Chapter 4), but any accompanying ΔSOC has not been added to national greenhouse gas accounts (MPIGA, 2008 p117).

The mature eucalypts in primary mixed-forests have large and dynamic humus mounds, nestled in the buttress region. Humus mounds are more voluminous around the larger *E. regnans* trees because of high fall rates of branches and bark (trapping further debris),

and high stemflow of rainwater. The widely spaced, mature eucalypt trees intercept oblique rain, from 35–70 m above the underlying understorey trees. The high stemflow results in part from the large crown volume, steep branch-inclination angle ~45º ((Figures 1-5, 5-1), and smooth bark (Crockford and Richardson, 1990). The large humus mounds are in part a product of the ecological integration of stemflow and the facultative epiphytes sensu Moffett (2000); or 'hemi-epiphytes' sensu Petrie et al. (1929), which can include mature trees (Levia and Frost, 2003; Oyarzún et al., 2011) (Figures 1-5, 5-1). The humus mounds can be up to 'several feet' deep (Cremer, 1962), increasing with tree age and size in the absence of fire (Ashton, 1981; Bens et al., 2006; Penne et al., 2010). The areas of increased stemflow infiltration are possible foci of microbial activity and diversity (Levia et al., 2012). The humic material contributes to SOC in the mineral soil below the humus mound, transported by canopy throughfall and stemflow (Liski, 1995) and therefore the humus mounds are themselves possible locations of concentrated C accumulation and are therefore potentially relevant to carbon accounting. The humus mounds and hemi-epiphytes complicate measurement of the already complex buttress shape and lower trunks of the large trees, and therefore also of volume and carbon content of the host tree.

Destructive sampling provides direct measurement of carbon mass but it initiates emission from that mass and for rare trees it violates conservation goals. Instead use can be made of both intact trees and those in industrial zones then geometric modelling can be used to derive parameters for the humus volume, the lower trunk, and roots.

The shape of the buttress regions of mature *E. regnans* has previously been measured using standard tape-measures but with relatively few measuring points for a curved surface of such complex curvature (Dean et al., 2003; Dean and Roxburgh, 2006; Sillett et al., 2015). Solid objects can be reconstructed from photographs taken with digital cameras, photogrammetric techniques, and particularly the structure-from-motion

Figure 5-1. (a) and **(b)** Macroscopic aboveground tree architecture. Acute angles, averaging near 45º, where large branches meet trunks of mature *Eucalyptus regnans* trees; see also Figure 1-5. Epicormic shoots have become large branches, forming lower crowns following stand self-thinning, crown volume

has increased (Styx and Florentine Valleys respectively, both trees demolished by logging). Tree '(b)' was 'El Grande' DBH= 6.38 m, height= 75.4 m, photographed during logging. **(c)** Sapling sassafras as hemi-epiphyte on *E. regnans*, joint to 3.9 m height above solum, Styx Valley. Tree is 'Chapel Tree', DBH= 6.03, height= 80.1 m (same tree in Figure 6-4.b, 6-6.a). **(d)** Most epiphytes cut away, person stood on epiphyte roots, prior to logging, DBH= 4.95 m (same tree in Figure 5-4).

multi-view-stereopsis algorithms employed by software such as Photoscan (Agisoft, 2015). The process is suitable for use with trees (Chapter 1, section 1.3.4). A digital camera and Photoscan were used in the present study.

Given the substantial knowledge gap, terrestrial photography, photogrammetric image processing and 3D modelling were undertaken in order to facilitate assessment of potentially significant carbon pools associated with large trees in a primary forest. The 3D-shape of the buttress region and roots was characterised, and those measurements were connected to standard biomass allometrics for use in carbon accounting and carbon dynamics at the stand-level and in LUC research. The formulae developed were applied to stand-level data in a companion study (Chapter 6). The experience with the technology is utilised to make some observations on its relative utility in comparison to measurement using LiDAR or tapes. The project is idiographic, rather than nomothetic, science, designed to describe unknown important relationships in nature, rather than to test hypotheses derived from theory.

5.3 Methods

The terms 'humus mound', 'buttress region' and 'footprint' are explained in the glossary (Appendix I).

5.3.1 Study site

Data were collected in primary tall-eucalypt forests with a rainforest understory (mixed-forest) in the Styx, Tyenna, Florentine and Kermandie catchments in Tasmania, Australia, across an elevation range of 221–659 m (Figure 5-2). Aboveground soil data for the buttress region were collected in the Styx Valley, where annual rainfall is 1170 mm (over the period 1993–2013) with a weak winter maximum. The mean daily minimum temperature per month is 2.3^{o} C and the mean daily maximum per month is 22.4^{o} C. The soils were cambisols derived from sedimentary rocks.

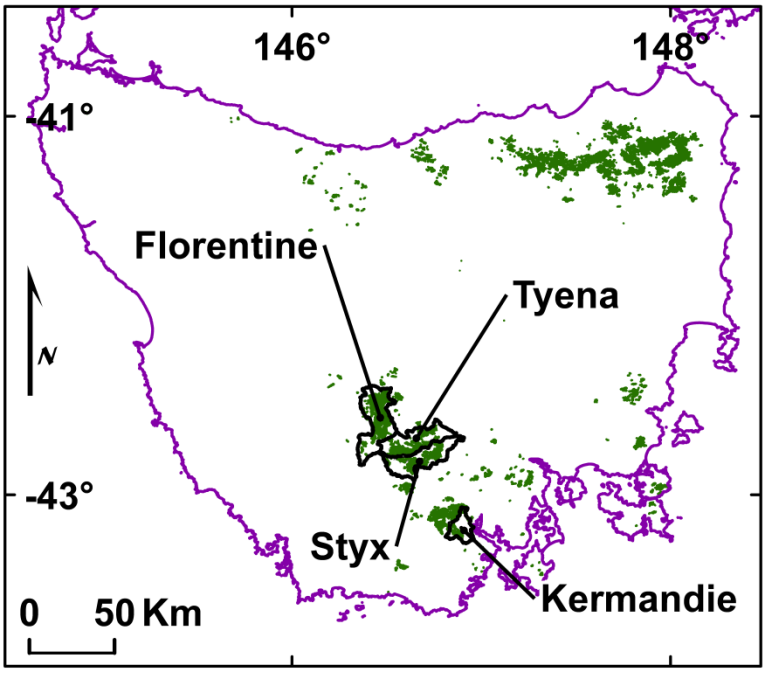


Figure 5-2. Study area. Catchments (outlined in black) in Tasmania where data were acquired. Green area shows approximate distribution of *Eucalyptus regnans*-dominated forests. Geographic projection, degrees latitude and longitude.

5.3.2 Overview of mensuration and modelling

Three dimensional models of twenty-nine large (mean DBH= 4.46 m, range 2.83–7.16 m)), in situ eucalypts were made using digital-SLR photography and Photoscan (Agisoft, 2015, versions 1.0.4.1847 to 1.2.0.2198), and were analysed using Photoscan and ArcGIS (Figures 5-3–5-7). There were 25 *Eucalyptus regnans*, 2 *E. obliqua*, 1 *E. regnans-E. obliqua* hybrid, and one was either *E. regnans* or *E. obliqua* (identification difficult as it had been logged and burnt). Samples were selected to cover a range of allometrics (including the high end) and to represent the prominent components of tree architecture and related ecology. Ground slope averaged 8 degrees (range 0–24 degrees). There were 20 live trees and nine cut stumps from logging, 14 measurable hollows and 20 undisturbed humus mounds. Four live trees had large basal fissures through which one could walk ('walk-in' trees). Similarly measured were under ten examples each of eucalypt roots, loose logging debris, myrtle trees (*Nothofagus cunninghamii*) (nine), sassafras trees (*Atherosperma moschatum*) (one), fallen trees, branches, root and trunk slices, and soil profiles. The myrtle and sassafras trees had no measurable humus mounds of their own. Measurements in 2D from three eucalypt buttress 'logs' with hollows, in logging debris, were added to the data set.

5.3.3 Photogrammetric procedure

For each object to be modelled, multiple, overlapping photographs were acquired over the whole region of interest, taken mostly perpendicular to the different surface sections. For each eucalypt tree, between 143 and 846 photos (average 375, range 143–846) were acquired of the buttress regions and partial upper trunks with a digital-SLR camera, from ground-level. The camera principally used was a 24.1 MP Nikon D3200 and AF-S Nikkor 18-200 mm lens with image stabilisation. For one eucalypt tree a different camera was used: a Pentax 14.5 MP K20D and Pentax DA 18-55 mm lens. For

photography for use in Photoscan the zoom lenses were fixed at minimum zoom. The aptitude of program Photoscan and its relevant photogrammetric background are introduced in section 1.3.4. The photography of large stumps and root balls required climbing over them, and taking photographs while ascending and descending in order to connect the models derived from different vantage points. The photographic technique followed recommendations in the Photoscan manual (Agisoft, 2015). The Nikon camera had a Solmeta GPS (Geotagger Pro 2) attached for geotagging the file headers of photos, recording routes to trees and to aid in photo orientation if necessary.

Prior to photography for 3D modelling, photogrammetric targets were placed on opposite sides of the object to be modelled, e.g. the buttress region of trees. These photogrammetric targets were of two types: 1) control points (referred to as ground control points– GCPs) which provided metric information that was used to control the scale of the photogrammetric model, and 2) tie points, which were not measured in the field but were well-defined, unambiguous targets observable in multiple photographs (Figure 5-3). Tie points improve the strength of the photogrammetric solution. Metric (object scaling to metres) control was established by including in the photography two 1.5 m long, light-weight bars, each of which had two wood screws drilled through, 1.200 m apart. The screws served three critical purposes: allowing the stake to grip the side of the tree, providing GCPs for the subsequent photogrammetry, and acting as scale marks. The heads of the screws were left proud of the stake and flagging-tape so that the junction of the axis of the screw and surface of the bar could be estimated during image processing in Photoscan. That junction was taken as the GCP— two per stake, from the two screws. During image processing the GCPs on each scale bar were located to the nearest ~0.5 mm, by visualisation, of the uppermost screw-stake join. Some larger models required the placement of the scale bars in more than one location in the field, and therefore merging of sub-models during image processing using common tie points.

Photogrammetric tie points positioned in the forest were mostly small, differently-coloured plastic spheres of contrasting colours and size, from 3 to 15 mm diameter fixed

to the surface using a nail (Figure 5-3). The colours were crucial to identification and differentiation on photographs, taken with either sky or vegetation in the background, and in different lighting. The visualised centre of the sphere was used as the tie point. The modelling process regularly necessitated further tie points to be added in order to complete photo alignment; these were opportunistically chosen from conspicuous, small features appearing in several photos. Such tie points were on field-equipment, or any natural or semi-natural object that had a sharp definition point and was observable from several angles, such as intersecting cracks on a cut stump. The average number of photogrammetric targets used per model was 15 (range 4–37).

The photos were aligned accurately, most often starting from a good approximation achieved automatically by Photoscan, followed by accurate manual identification of each GCP and tie point on each photograph (wherever they were visible). The latter process was iterative as the initial automated placement of new tie points improved as the number of correct camera positions increased. Each alignment run typically took 24 hours.

Quality control of the accuracy and precision of the modelling was achieved by correcting photo alignment using as many tie points as necessary, to ensure that there was only one sensible model fragment to which all photographs contributed. Blurred photos and photos with an insufficient number of image areas to match with neighbouring photos, were not used. An indication of the quality of the solution was the derived length of each of the two scales bars. The average error between scale bars in processed imagery was 2.3 mm (range 1.4–7.7 mm), i.e. a relative distance error on one side of the 3D-model compared with the other of 0.19%, range 0.011–0.64%).

Each model improvement step in Photoscan consisted of photo-alignment, dense point cloud generation and mesh generation, which typically took ~24 hours. The number of facets (indivisible planar surface elements) for final model construction was set at 5,200,000. Complete processing took about 1.5 weeks per model on a Windows PC,

with 8x Intel Core i7-3820 3.60 GHz processors, 64 Gb RAM and a NVIDIA Geforce GTX 670 graphics card.

Once the photos had been correctly aligned and maximum accuracy and precision achieved, models were oriented to their natural vertical orientation using a combination of thin, background trees (vertical) and the diameter-tape (horizontal). Vertical translation was performed to place the average diameter-tape location at 1.3 m above the top of the mineral soil which was given the height of 0 m (in accordance with the forestry standard used during fieldwork).

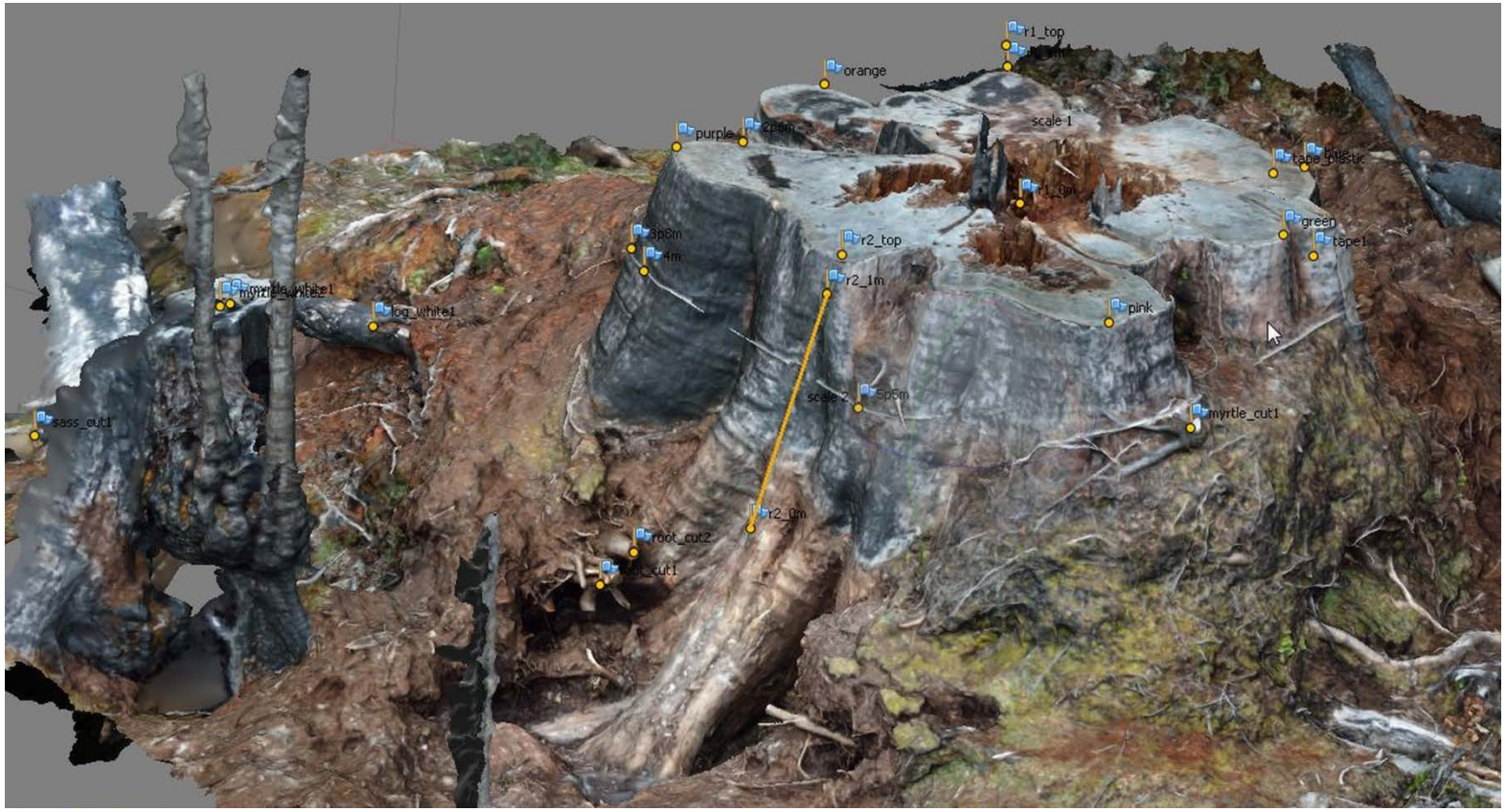
The model data were exported in 3D colour format (.wrl, .pdf and .obj), and 2D format for standard views: GeoTIFF (orthophoto with 2 mm object space resolution) and DEM. The 3D formats were used in visualisation to record or reveal ecological aspects, and the 2D formats were used for shape analysis in ArcGIS. The 3D models also provide a data archive for future use (such as stem taper and moss area) in case the trees are destroyed by logging or fire.

The models' orthophotos and DEMs were imported into ArcGIS. Geometrical attributes determined in the GIS were DBH (after correction for humus, large epiphytes, burls, CWD and the diameter tape), cross-sectional area (enveloping area including bark), hollow area, footprint area, ground slope; nd humus mound area and volume (Figure 5-4). Equations of simple form and high R^2, correlating geometric attributes of tree architecture, were selected from the range of analytical models suggested by Eureqa (Schmidt and Lipson, 2009) and parameters were refined by non-linear regression in LABFit (da Silva and da Silva, 2015), which uses the Levenberg-Marquat algorithm for least-squares, optimising the sum of squared errors in y. When selecting equations from Eureqa output a compromise was made by selecting those of higher R^2 but still not too complex, i.e. as few parameters as possible and where gains by small decreases of R^2 were no longer sufficient to warrant extra parameters. It was a tuning compromise process learned by experience.

5-3(a)


purple
euc_crown1
sass_crown1
r2_top
r2_1m
pink
green
blue
3p8m
4m
myrtle_cut1

5-3(b)



5-3(c)

Figure 5-3. Example ground control points and tie points used. DBH= 3.11 m. **(a)** Placement of GCP at the intersection of screw and rod at top of scale bar, and tie point on end of rod. **(b)** Tie points on distant and near objects. **(c)** Completed 3D model with large number of tie points needed for object's complexity, variety of backgrounds [in photos], and range of ground elevation from top to bottom. (Same tree in Figures 6-3 and 6-4.d.

Hollow area and cross-sectional area were determined as functions of DBH, footprint was determined as a function of DBH and slope, humus dimensions were determined as a function of footprint. Double and floating point precisions were used in calculations throughout, so as to minimise accumulation of rounding errors. The number of significant digits written in the thesis is kept near five in order to prevent accumulation of rounding errors during successive calculations by readers of the work.

5.3.4 Retrieval of buttress-region data using GIS

It was necessary to correct the field-measured DBH, due to possible incorrect placement of the tape measure around such large girths, (up to 22.5 m). This was achieved by GIS analysis of the 3D models (using ArcGIS). Contours at 0.1 m intervals were created in the GIS, including one at the height of the DBH tape (1.3 m). That 1.3 m-level contour was corrected ArcGIS for humus on top of the bark, large epiphytic trees, burls, and for the diameter tape itself, which the Photoscan solution sometimes included as part of the model, near buttress spurs. The colours in imagery were crucial to differentiating material types, for example bark from moss. When the 1.3 m-level contour was thus adjusted a convex-hull polygon was calculated that modelled the ideal placement of a DBH tape, its perimeter giving a more accurate measure of tree DBH. The DBH of one *E. regnans* could not be measured in the field due to proximity of a mature epiphytic myrtle, with both trunks joint up to ~5.4 m. Instead the DBH was estimated using the GIS from a cross-section of the 3D model, with subtraction of the epiphyte (Figure 5-7). The measurements in the GIS allowed the area enclosed by bark at 1.3 m to be determined and the hollow area for stumps and walk-in trees. Data from three buttress logs in logging debris were included in the hollow calculation. Formulae for hollow area and bark-enclosed area were determined as functions of DBH.

The trees' footprint (see glossary) and humus mound dimensions were also measured using GIS. A rough sketch of the footprint was provided where the contour lines no longer showed bumps due to the lateral roots. It was refined by on-screen digitising. Ground slope was also measured in the GIS between the highest and lowest points just outside of the footprint. An equation for footprint was determined as a function of DBH and slope using the program Eureqa (Schmidt and Lipson, 2009).

Humus volume was determined by locating the area of humus (as seen in vertical projection) in the flutes between spurs using the GIS (Figure 5-4). The areas were delineated by digitising the representative small polygons between the bark of the flute and the convex-hull polygon at 1.3 m height (representing the ideal DBH tape placement). The slope of the surface of the humus was determined by drawing radiating lines from the bark pointing away from the centre of the tree to the forest floor, and overlying the centre of the humus filling the buttress flute. The lines were projected onto the humus surface using ArcGIS 3D-Analyst. The vertical profiles of the lines were then exported to MS-Excel and corrected for any bumps due to debris such as fallen branches etc. Of the 29 eucalypts modelled, 20 were suitable for determination of humus volume. The humus mounds of the others eucalypts were either: substantially burnt (and therefore vaporised), mechanically disturbed during logging, too concealed by vegetation, or they had been dug away to allow access to the 1.3 m level for DBH measurement.

The humus mounds had linear slopes (height/length= -0.7826(0.2366), i.e. ~38.0(8.5)°, N= 20; for line of best fit for individual trees $R^2 = 0.98$). This simplified the determination of humus volume. Although the humus mound in a flute spreads sideways further from the trunk and meets humus from neighbouring flutes, its area was approximated as two parallel lines perpendicular to the edge of the convex-hull at 1.3 m height bordering the flute (Figure 5-4). Another approximation made was that the flute area containing humus could be represented as a triangle (of the same area, and with the same base length on the convex-hull, as the irregularly shaped flute). The sides of each

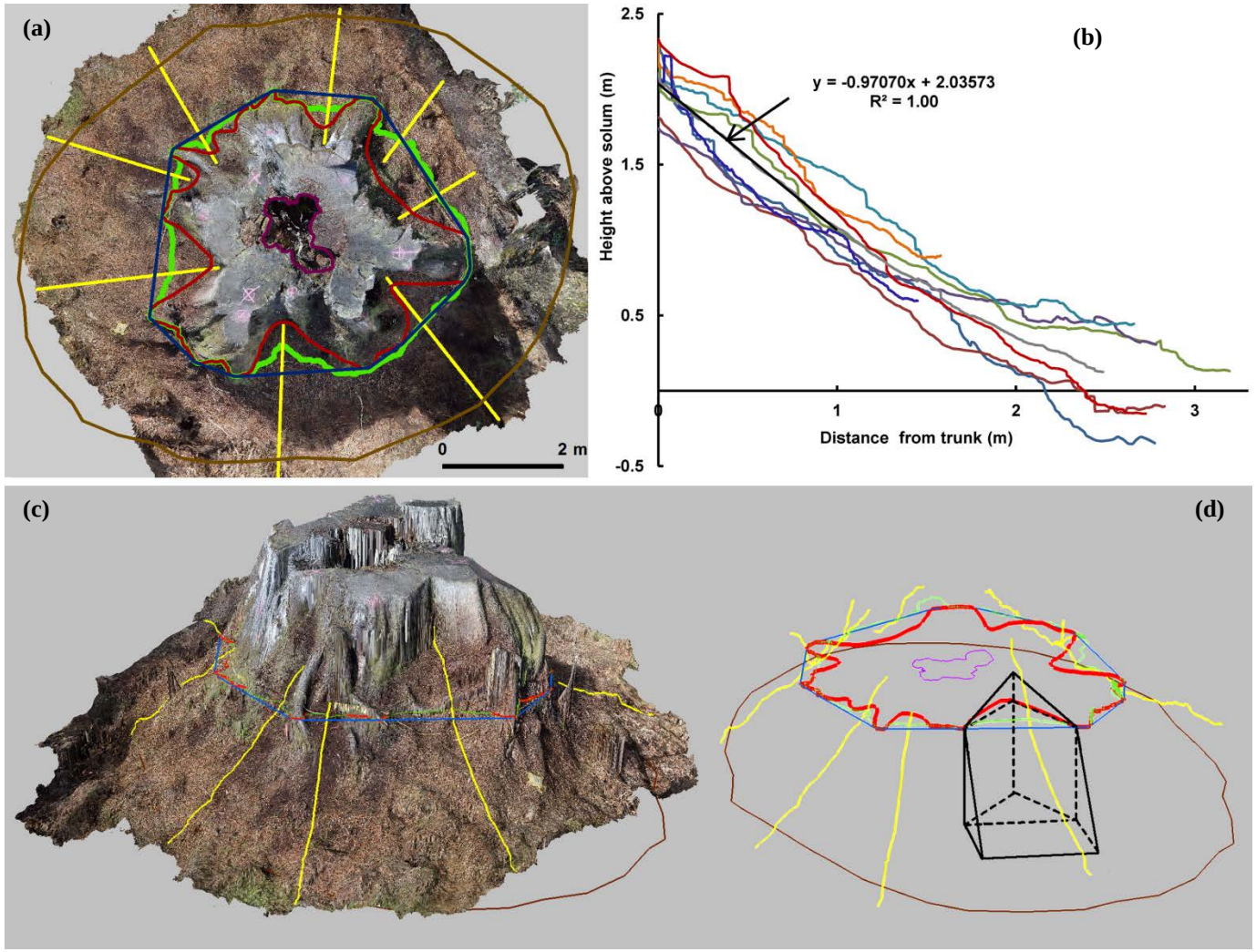
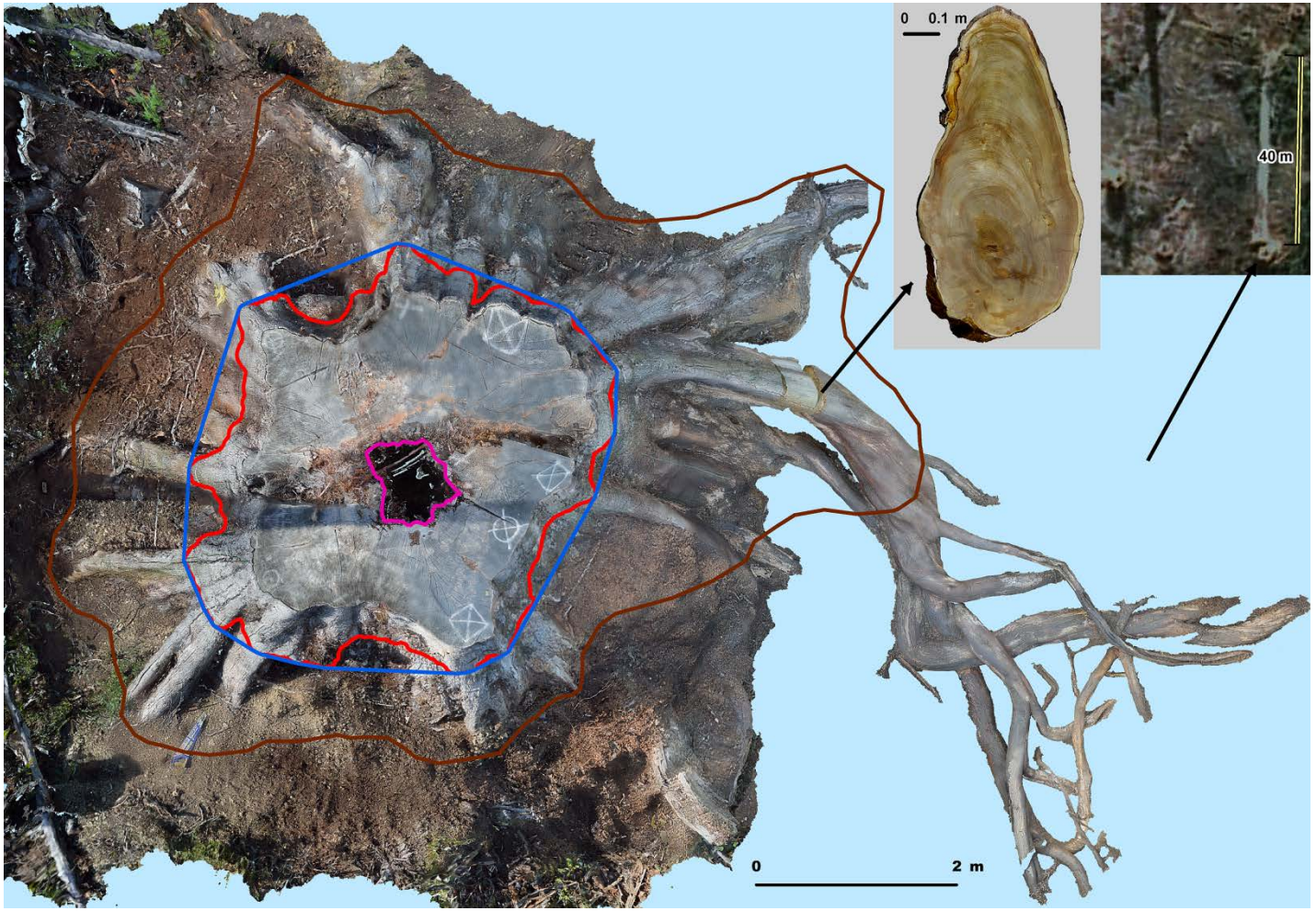


Figure 5-4. 3D-model from Photoscan with ArcGIS markup, showing buttress region and humus mound. *Eucalyptus regnans*, DBH= 4.95 m, shown before logging in Dean (2003) and Figure 5-1.d. **(a)** Top view, lines: outermost= footprint, innermost= hollow, in between and convoluted= corrected 1.3 m contour, moderately convoluted= 1.3 m contour, least convoluted= convex hull at 1.3 m, radial= humus profiles. **(c)** Oblique view, example triangular and rectangular pyramids used to calculate humus volume for a flute.



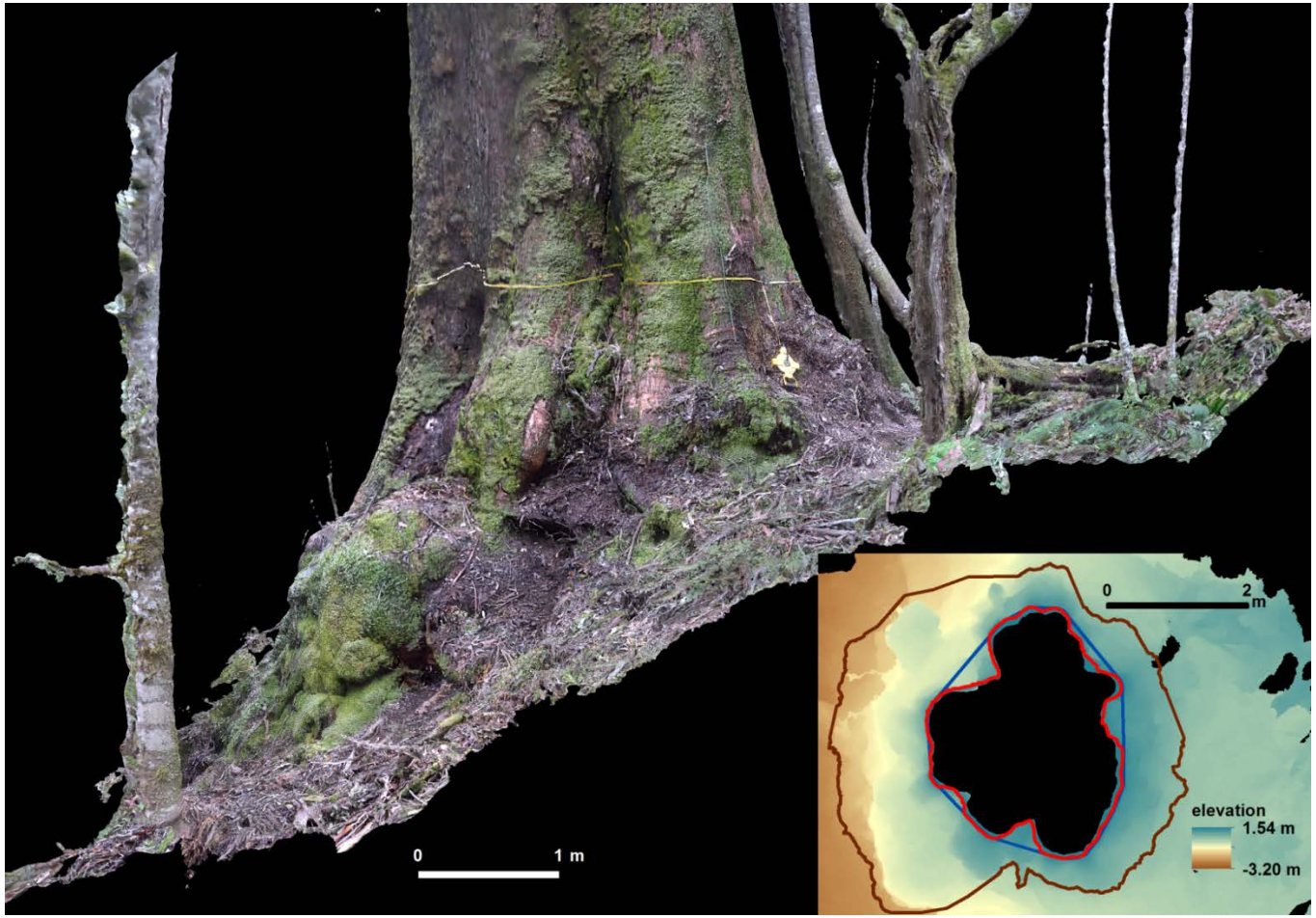
(a)

Figure 5-5. Buttress area characterisation from 3D model, example 1.

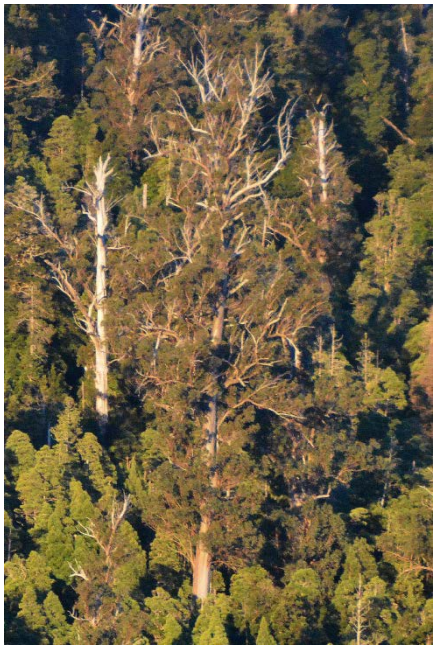
(a) Orthophoto created using terrestrial photography and Photoscan, top view of a *Eucalyptus regnans*, (DBH 4.38 m, Tyenna Valley, coupe TN050E, Figure 6-4.a). Red line= 1.3 m corrected contour, blue line=convex hull at 1.3m matching DBH tape, brown line=footprint, magenta line=hollow. Humus mound removed, soil removed from one major lateral root on downhill side. Coarse roots extend beyond footprint and typically limit soil sampling. Roots continue spiral grain of trunk and are plaited, curling around hemi-epiphytic trees and creating stability against root slip and tree fall, facilitating great height of the species. **(a)** Left hand inset & **(b)**: 3D model of root slice of large lateral within the footprint, ring age count= 350(±40) years: most expansive growth on top side, corresponding to buttress width expansion. **(a)**-Right hand inset & **(c)** (GoogleEarth ®) satellite image shows felled trunk, stump, and neighbouring stumps, during logging (scale bar = 40 m).

(c)

(b)

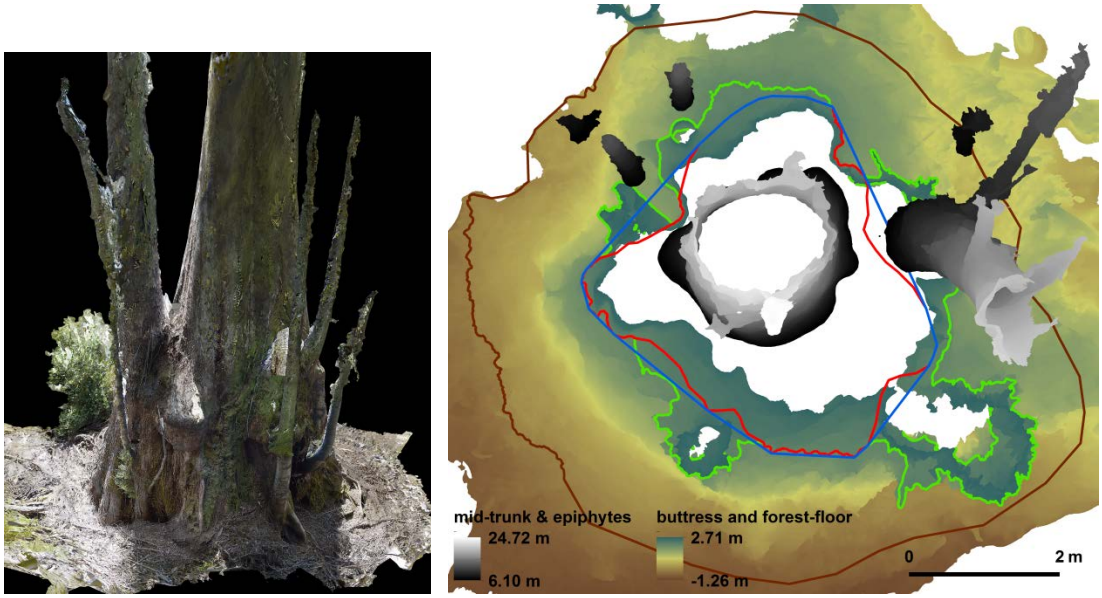


(a)



(b)

Figure 5-6. Buttress area characterisation from 3D model, example 2. **(a)** orthophoto created using terrestrial photography and Photoscan. *E. regnans* (DBH 3.24 m, height 64 m, Styx Valley) on steep slope (24º) with minimal humus mound: volume=0.96 m^3. Inset shows topography (DEM), 1.3 m corrected contour, convex-hull and footprint. **(b)** upper portion of tree in '(a)', above neighbouring rainforest understorey, and with ample foliage and original crown.



(a) **(b)**

Figure 5-7. Wider use of 3D models. Model of *E. regnans* DBH= 4.56 m (coupe SX009C), on left in Figure 1-5. The DBH could not be measured in the field due to the large hemi-epiphytic myrtle (LHS in **(a)**, RHS in **(b)**). The myrtle could be separated in ArcGIS using cross-sections and contour levels, and the DBH then estimated, as shown in (b). (Green line= 1.3m contour, red line= corrected 1.3 m contour, blue line= 1.3 m convex hull, brown line= footprint). The lower, oblique view of the model is as viewed from uphill in the top view.

flute's humus were approximated as being vertical. Thus there were two volumes to calculate per flute: a triangular prism with a sloping top, and a rectangular prism with a sloping top. In some instances there was no humus (e.g. the large flute on the entry side of a walk-in tree) or the humus mound did not reach the convex-hull of the diameter tape (in which case there was only the triangular prism to estimate). The parallel-sides approximation of the rectangular prism part of flute-humus, which underestimated the amount of humus, was intended to compensate for the approximation of a vertically sided flute, which would generally overestimate the amount of humus as the sides of the buttress roots often spread inwards below the surface of a deep humus collection.

5.3.5 Subtraction of coarse root volume near trees

5.3.5.1 Generic tree species

The below-ground root volumes in the buttress region of trees were estimated— the 'effective root volume'— in order to adjust the typically measured soil bulk density and SOC per unit-area for the soil displaced by those roots. That calculation required root volume to be formulated as a function of DBH (as it is a common measurement) and of depth. For that purpose a combination of the 3D-modelling, tree packing geometry, data from the literature (Ashton, 1975a; Nicoll et al., 2006) and formulae from Chapter 3 was used.

As a first approximation, and the for the default value in the absence of other evidence, a medium rate of 2.8610 m^{-1} for the exponential falloff of root volume with depth was chosen from the literature. This value was for 40 year old Sitka spruce on sloping terrain (Nicoll et al., 2006 their Fig. 5). Determination of the falloff rate under tree trunks requires excavation and consequently literature data for the phenomenon are sparse. This age of tree was considered appropriate because it has a smaller lateral root volume than a mature tree and thus represents more the central region under a mature tree, which is the region of interest in the present work. The data presented in Schwarz et al. (2010 their Fig. 4) fitted an exponential falloff with a rate of 4.4889 m^{-1} but were for the whole root systems of boreal and cool temperate forests, probably of various ages. A lower falloff rate has been recorded: ~1.5 m^{-1} (Eamus et al., 2002), probably due to a lower moisture-balance environment (Canadell et al., 1996).

A single root falloff rate is an approximation for the combined falloff rates of tap, sinker and lateral roots; and the rate below the buttress region is more dependent on taproot or sinker roots than it is for the root system as a whole. The SOC falloff rate used was from data for the Styx Valley (Dietrich, 2012). The percent reduction in SOC per unit-area was determined by dividing the soil and the root volume into fine depth increments,

subtracting the soil volume corresponding to root volume, then recalculating the SOC per unit-area, and comparing with the original SOC per unit-area.

To determine the pathway for further development of the methodology, a sensitivity analysis of all parameters (falloff rates, root mass and depth increment) was performed using stand-level root data for oldgrowth Douglas-fir (Santantonio et al., 1977). The stand-level data for oldgrowth Douglas Fir (Santantonio et al., 1977) were combined with the published SOC data for the Styx region (Dietrich, 2012) and the root volume falloff rate described above (Nicoll et al., 2006 their Fig. 5). Equations representing trends were found using Eureqa and the simplest solutions were chosen that had R^2= 1 within five significant digits. The sensitivity analysis showed four major trends that were helpful in further modelling and for stand-level calculations:

1) The percentage SOC reduction at the stand-level when accounting for root volume decreased with step size and changed in the third significant figure if vertical step size was >0.05 m (while keeping root volume constant). All further calculations were done with a 0.001 m step.

$$SOC\_reduction\% = 4.5778 - (1.0970 step^{1.6506}) \quad \text{Eq5-1}$$

2) The percentage change in SOC per hectare was positively correlated to the root-volume-falloff-exponent. (I.e. shallower-rooted trees decreased the SOC per unit area more than deeply rooted trees, for the same total root volume, which is logical as SOC density is higher nearer the surface.)

$$SOC\_reduction\% = 6.86556 root\_falloff\_rate/(1.4152 + root\_falloff\_rate) \quad \text{Eq5-2}$$

3) The percentage change in SOC per hectare was linearly proportional to effective root volume per hectare.

$$SOC\_reduction\% = 0.0074888 effective\_root\_volume \quad \text{Eq5-3}$$

4) The percentage change in SOC per hectare increases with the *SOC_falloff_rate* with depth (while keeping root volume constant) but with an asymptote of ~13.7%. It was approximately linear within the bounds of typical SOC falloff rates (i.e. 1–2). This simplified the calculation of different root-volume effects for different SOC-falloff-rates.

$$SOC\_reduction\% = 13.717 - 39.781/(2.9124 + SOC\_falloff\_rate) \quad \text{Eq5-4}$$

5.3.5.2 *Eucalyptus* roots

Typical root architecture and volumes were estimated from Ashton (1975a), from observations of the root architecture of numerous trees upturned during logging or by logging plus windthrow, and from 3D-modelling of the buttress and roots of five logged trees, using Photoscan. In summary, as the *E. regnans* tree matures the central tap root or tap roots decompose and are replaced by a separate sinker root adjoining each major lateral root, with these laterals forming the spurs of buttresses. There are sometimes much smaller sinker roots further along each lateral, within the footprint.

From the present observations the width of *E. regnans* sinker roots was ~$^1/_3$ that of lateral roots. The 'summary sinker' is all sinkers added together for a given lateral (Figure 5-8). The lateral and sinker roots were modelled as cones, one summary sinker per lateral and 20 laterals per mature eucalypt tree. The choice of 20 laterals, each with one sinker root, followed observations of mature and senescing-mature *E. regnans* trees

(Ashton (1975a) and the present work), combined with approximations necessary for mathematical modelling. Horizontally, these laterals began at a distance of ~DBH/2 from the tree centre. The length of the laterals was held constant at 11.3-(DBH/2) m, with the value of 11.3 obtained from Ashton (1975a) (9–13.6 m). Additionally, 11.34 m is the radius of packing non-overlapping circles (not that trees were assumed to be regularly spaced and free of root overlap), each representing a *E. regnans* tree in a 400 year old stand occupying one hectare, assuming the stand density of 18.61 trees ha^{-1} from formulae in CAR4D (Dean et al., 2003). It is also the radius of equivalent circles representing the non-overlapping, space-filling area occupied by a stand of 320 year old trees of ~25 ha^{-1}, i.e. close to the 27 ha^{-1} for a densely-packed stand >300 year old (Gilbert, 1959). Inter-tree distances for *E. regnans* are of course not regularly arranged (Jarrett and Petrie (1929) and observation of juxtaposed trees in the present study), but it is expected that root overlap would be minimised where possible, as is canopy overlap, achieved during self-thinning.

(a)

(b)

Figure 5-8. Sinker roots on mature *E. regnans*. **(a)** Large sinker root beneath edge of buttress and adjoin the large lateral (right-hand side of photograph). The area around the tree had been logged (coupe SX009C) and the tree had fallen and its buttress split open. **(b)** The tree in coupe TN050E, Figure 5-5 had been logged. Soil from the coarse roots on one buttress spur was removed. Small sinker roots, not part of the buttress, were within the footprint zone.

The laterals are concentrated at 0.3 m depth (Ashton, 1975a) and therefore the diameter of the lateral from which sinkers develop at the start of the lateral was set at 0.6 m. The summary sinker roots were modelled as finishing at 2.55 m depth, from Ashton (1975a), and from observation in the present work of ~2.5 m long sinker coarse roots on upturned trees. The root-volume-falloff-rate of *E. regnans* was calculated for this approximate model. These approximations contrast with the precision of the buttress-region photogrammetry but the high precision of that work carries through to the stand-level calculations and the equations are available for use in other studies.

The total root volume for *E. regnans* was determined from formulas in program CAR4D (Dean et al., 2003) as a function of age, and the comprehensive output of annual stand attributes from CAR4D was reformulated as a function of DBH using Eureqa:

$$Er\_total\_root\_volume = 2.3326DBH^2 \quad \text{Eq5-5}$$

The affected region [where SOC is usually not measured near a tree] was assumed to have a radius 1.5 times that of a circle of equal area to each tree's footprint. This value was chosen subjectively, based on observations and modelling of bared roots, and in the absence of any other information. The root volume within this region, i.e. the 'effective root volume' (being part of the total root volume), was partitioned between the lateral and sinker roots, trimmed to the 1.5xfootprint area, split into 0.001 m vertical increments, and its effects summed over depth. The effect on SOC per unit-area, for a range of DBHs (1–8 m), was reformulated using Eureqa, for use at the stand level.

The *E. regnans*–dominated forest contains in places some *E. obliqua* and *E. delegatensis*. For these two species the same area around the trunk affecting SOC measurement as for *E. regnans* was assumed (1.5xfootprint where footprint was calculated as for *E. regnans*). Their root volumes were calculated using formulae for aboveground biomass and assuming 15% of total biomass was roots (Chapter 3), which was an average value 10.5–19 % (Carnahan, 1977; Feller, 1980; AUSLIG, 1990; Mokany et al., 2006). Wood densities were from Ilic et al. (2000). Effective root volumes were determined as a function of DBH, assuming the same geometry as for *E. regnans*, and using Eureqa to determine suitable functions for stand-level use.

5.3.5.3 Rainforest species roots

As a test of the assumed value for root:shoot ratio and root-falloff-rate, the aboveground and below-ground volume of one whole, mature myrtle of DBH 1.76 m and height 32(4)

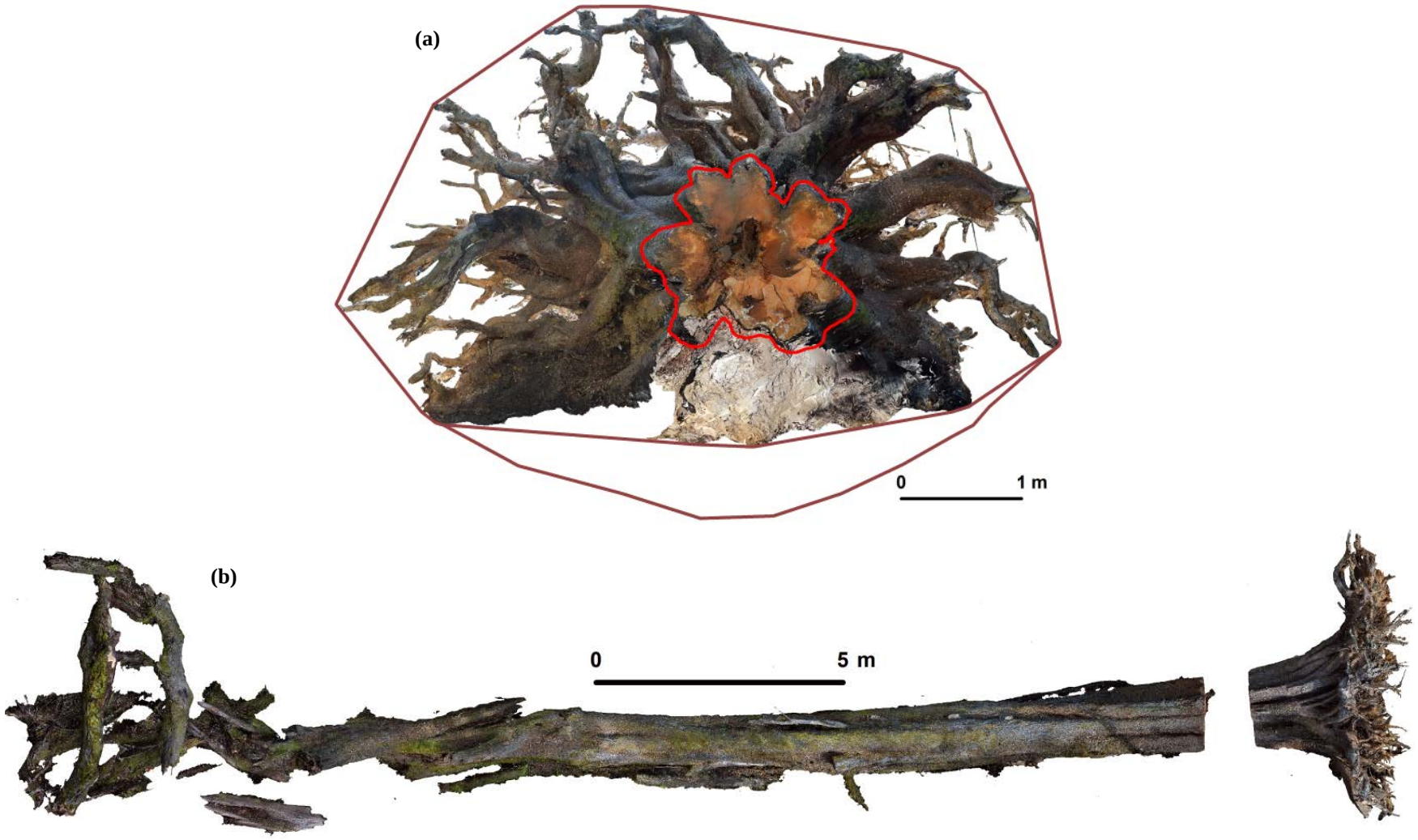
m, was measured. The trunk hollow was subtracted from the outer surface volume. It was a mature, senescent tree pushed over during logging. Its major lateral roots had snapped off at about the distance from the tree centre within which SOC is typically unmeasured— meaning that the observable roots constituted the 'effective root volume'.

Soil was carefully removed from the roots with hand tools (Figure 5-9). A 3D model of the root system and lower trunk was created from 356 photographs in Photoscan and reoriented to its original, upright position (Figure 5-10). Roots greater than ~0.01 m diameter were captured in the model. The model was divided into 16 horizontal segments, approximately 0.1 m deep, and the root volume was measured for each segment in the 3D-model using Photoscan. An exponential function for volume (i.e. effective root volume) as a function of depth was determined by nonlinear regression (Figure 5-10.d) and the effective root volume was 3.67 m^3.

The convex hull area of soil enclosed by the roots, as seen from above, was determined using GIS. The root model was merged with a second model, being that of the whole aboveground portion of the tree. The branches had been broken but the leaves and fine twigs had dried and fallen and therefore it was possible to measure the whole trunk and branch volumes by importing the DEM and orthophotograph of the 3D-model into the GIS. The volume of the whole tree was also calculated from the 3D-model from within Photoscan. The two volumes were not significantly different, within error margins, and their average was used in further calculations.

To determine the root:shoot ratio for the whole myrtle, major lateral roots were modelled as 10 m long cones. That distance corresponded to the distance observed where major lateral roots of myrtles tapered to ~0.005 m in diameter. Total root volume was 5.5 m^3, and effective root volume was therefore ~66.7% of total root volume. The dimensions of the pushed-over myrtle were: aboveground volume 30(4) m^3, total volume 36(10) m^3, fraction roots 15(4)%, root:shoot ratio 0.18(0.05).

Figure 5-9. Process of myrtle root excavation, photography with scale bars, and subsequent 3D-model. (Prof. Kirkpatrick in upper left.)


(a)
0
1 m
(b)
0
5 m

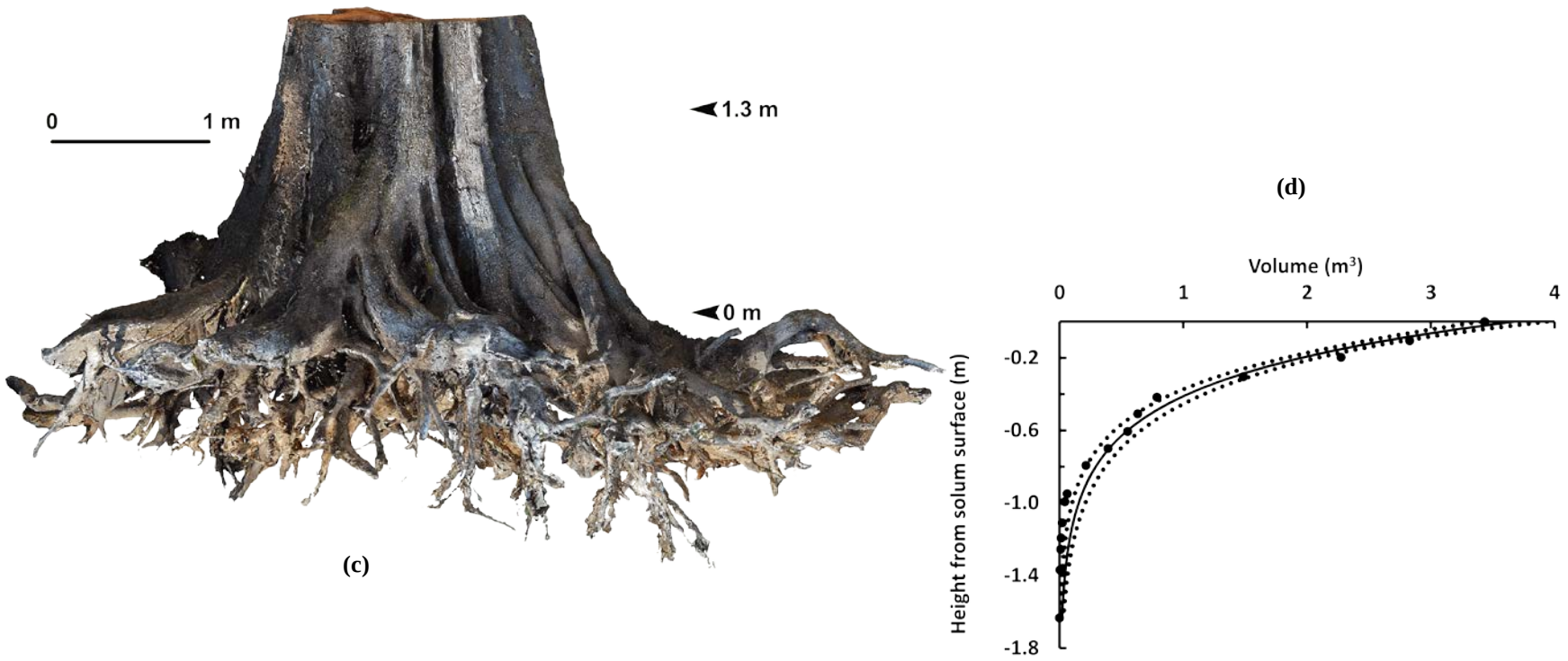


Figure 5-10. Components of geometric analysis of myrtle tree, from 3D-model made using Photoscan.

DBH 1.76 m, height 32(4) m, pushed over during logging. **(a)** orthophoto, top view showing cross section at 1.3 m (red outline) and convex hulls (visible and estimated total, brown lines). (**b)** orthophoto, top view of whole fallen tree. Gap on RHS was a section removed to measure trunk hollow, but it was included in the root:shoot ratio calculation. (**c)** orthophoto, side view of model showing root volume decrease with depth. **(d)** effective root volume as a function of height from the mineral soil surface.

The average length of coarse roots (to their point of snapping) was 2.59(0.39) m (N=42 points), as measured from the tree centre. The area of a circle enveloping roots of that radius was 21.1(3.1) m^2. The irregular shape projected by the roots, including an estimate of those not visible and still below ground, was 19.4 m^2. These two values are estimates of the area influenced by the effective root volume.

The empirical derivation above for effective root volume was compared with a theoretical value derived from the aboveground biomass for rainforest species given by Eq3-4. By combining that allometric with a wood density of 577.3 Kg m^{-3} (Ilic et al., 2000), and assuming root biomass is 15% of total biomass (Chapter 3), the estimated root volume for the pushed-over myrtle was 6.6 m^3. This was considered to be close to the empirical value of 5.5 m^3.

5.3.6 Soil sampling and analysis

Soil samples were collected from the outside of the humus mound of the buttress region with an Eijkelkamp© soil corer (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands) by augering, with and without digging pits. Eight samples in total were taken from 0.05 to 1.36 m aboveground.

The soil samples were air dried at room temperature (20º C). No stones were present. Roots >0.0005 m were removed from the samples, weighed and their volumes measured by water displacement. Soil volumes, bulk densities and carbon densities were adjusted for roots. Soil samples were ground in a mortar and pestle followed by grinding to a fine powder using a ball mill (Mark Hovenden, University of Tasmania, MM200 Mixer Mill, Retsch GmbH, Haan, Germany). The samples were analysed for total nitrogen, carbon and hydrogen using a Thermo Finnigan EA 1112 Series Flash Elemental Analyser, by Thomas Rodemann at the University of Tasmania. The low pH of the samples, recorded

in the field (<4.5), did not justify assaying for inorganic carbon. SOC density data for the different samples were not differentiable and were averaged.

5.4 Results

5.4.1 Aboveground buttress region of eucalypts

The average correction between the GIS-derived DBH measured from the 3D models and the field-measured DBH was -0.25% (-15–+7.6%). The data showed that there was a general but non-significant trend of underestimation of larger-tree DBH in the field and overestimation of the smaller trees. This suggests that for most trees, including the large ones, my method of DBH measurement in the field would not significantly overestimate carbon stock. The two largest corrections of -14% and -15% were due to burls and immovable logging debris respectively, and it was known during data collection that it would be necessary to use the 3D-model to provide corrections. The error from such trees could be reduced if it was known during fieldwork that no correction was available from data processing.

Parameters for equations are given in Table 5-1.

The footprint of eucalypt trees was given by (Figure 5-11.a):

$$footprint = DBH^2(a - slope) \quad \text{Eq5-6}$$

where *footprint* is in m^2, *DBH* is in m, and *slope* is tan of the ground slope angle. The negative correlation with slope concurs with Nicoll et al. (2006) where downhill lateral roots descend more quickly into the soil on steeper ground.

For reference purposes, if slope is not measured, and Eq5-6 cannot be used, then:

$$footprint = aDBH^2 \quad \text{Eq5-7}$$

where *footprint* is in m^2, *DBH* is in m.

The humus area within the humus mound, which envelopes the buttress region, and the humus volume above that area, were both functions of the footprint (Figure 5-11.b). Humus area within the humus mound was given by:

$$humus\_area = a\left(footprint\ b^{footprint}\right)^{1/2} \quad \text{Eq5-8}$$

where *humus_area* is in m^2, and *footprint* is in m^2 given by Eq5-6. Humus volume, above *humus_area*, was given by:

$$humus\_volume = a\ \exp(b\ footprint) \quad \text{Eq5-9}$$

where *humus_volume* is in m^3 and *footprint* is in m^2 given by Eq5-6. Large buttress roots occupied 49(15)% of the whole footprint, and humus area covered 40(19)% of the area between the projected stem basal area and the footprint perimeter.

5.4.2 Carbon density and cumulative SOC

The soil bulk density in the humus mound was 279(125) Kg m^{-3}, SOC density was 91(13) Kg m^{-3}, C wt% 37(11), C/N ratio 34(7), and H_2O wt% 57(14). Applying the carbon density to the equation for humus volume (Eq5-9) gives the equation for SOC in an individual tree's humus mound as:

$$humus\_SOC = a\ humus\_volume \quad \text{Eq5-10}$$

where *humus_SOC* is in Kg and *humus_volume* is in m^3 as given by Eq5-9, and *a* is the humus mound SOC density. Thus for the larger eucalypts measured (DBH> 5 m) on flat ground there may be ~1.5 tonnes of C in the humus mound.

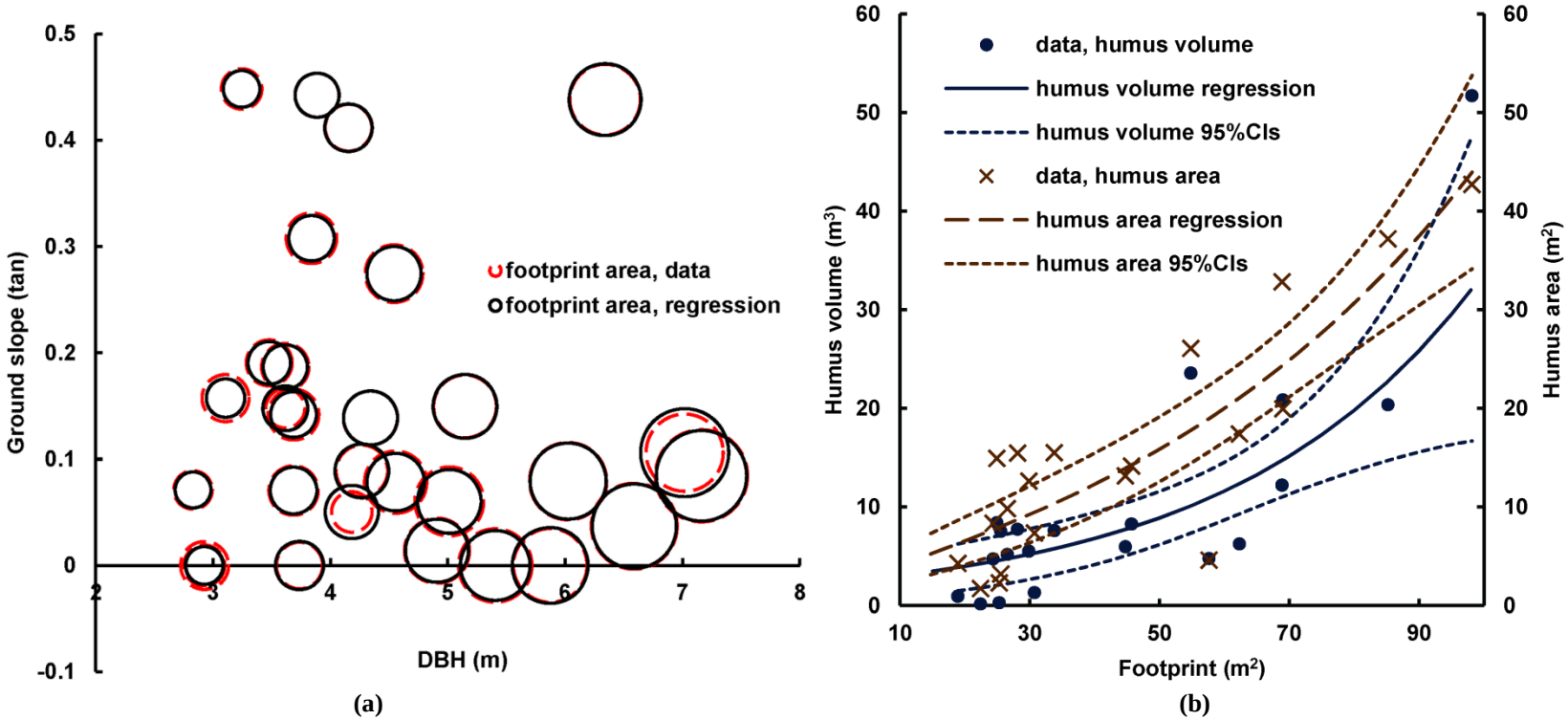


Figure 5-11. Some attributes correlated with individual tree measurements.

(a) Footprint of eucalypt trees (represented by circle size) versus DBH and ground slope. The 95% confidence intervals, corresponding to 2.6% of the one parameter, were too narrow to show at this scale. **(b)** Humus-area within the humus mound of eucalypt trees and the corresponding humus volume, as functions of footprint.

Table 5-1. Parameters for equations derived from empirical data.

Equation	a	SD	units	P(t)	b	SD	units	P(t)	df	R^2	*P*
Eq5-6	1.93190	0.0496554	none	<0.0005					28	0.88	<0.005
Eq5-7	1.78703	0.0631358	none	<0.0005					28	0.85	<0.005
Eq5-8	1.10367	0.25471	m	<0.0005	1.02874	0.00624	none	<0.0005	19	0.76 adj.	<0.005
Eq5-9	2.33694	0.91439	m^3	0.02	0.0267006	0.0057880	m^{-2}	<0.0005	18	0.52 adj.	<0.005
Eq5-10	91	13	Kg m^{-3}	N/A						N/A	
Eq5-14	3.67546	0.123688	m^3	<0.0005	3.16199	0.17464	m^{-1}	<0.0005	14	0.98 adj.	<0.005
Eq5-16	0.634	0.011	none	<0.0005					31	0.97 adj.	<0.0005
Eq5-17	1.2057	0.5323	m	0.043	-3.1676	2.3692	m^2	0.206	12	0.24 adj.	0.04
Eq5-18	0.28440	0.03092	none	<0.0005	3.5949	0.82143	none	0.001	19	0.74 adj.	0.0002

5.4.3 Effective root volume

Effective root volume was a species-dependent function of DBH. The root volume falloff rate for the eucalypt model with conic frustum laterals and conic sinkers was 3.133 m^{-1}, and that for the single myrtle empirical examination was 3.182 m^{-1}. These were considered similar (within experimental error) to the value of 2.8610 m^{-1} from Nicoll et al. (2006). For eucalypt roots the modelled value could be used, but as only one myrtle was examined experimentally there was insufficient reason not use the value of 2.8610 m^{-1}, for non-eucalypt species. The linear proportionality between effective root volume per hectare and effect on SOC per hectare (Eq5-3) simplifies stand-level calculations in the tallying of root volume effects for individual forest plots. With the slightly higher *root_falloff_rate* of 3.133 m^{-1} for eucalypts, the parameter in Eq5-3 increases to 0.0098756. For eucalypts in a stand, equations Eq5-11, Eq5-12, Eq5-13 (*E. regnans*, *E. delegatensis* and *E. obliqua* respectively) and Eq5-3 can be used to tally the total effective root volume per hectare and the corresponding percentage reduction in SOC due to the roots below the buttress region:

$$Er\_effective\_root\_volume =$$

$$0.93654DBH^{1.2934} + 299.19\exp(-12.611/DBH) \qquad \text{Eq5-11}$$

$$Ed\_effective\_root\_volume = 40.0675DBH/\exp(7.2377/DBH) \qquad \text{Eq5-12}$$

$$Eo\_effective\_root\_volume = 1.8253DBH^{2.3359} \qquad \text{Eq5-13}$$

These equations appear a little complex compared with those for humus and footprint but the effective root volume is a subset of the entire root volume, with its measurement boundary chosen for pragmatic reasons. For the example myrtle tree, from modelling the effective root volume as a function of depth:

$$understorey\_effective\_root\_volume = a\mathrm{e}^{-bz} \qquad \text{Eq5-14}$$

where *understorey_effective_root_volume* is volume in m^3 and z is the distance from the mineral soil surface in m (parameter values in Table 5-1). For understorey trees

with DBH ≥1 m, when doing stand-level calculations, effective root volume can be calculated assuming linear proportionality with DBH^2 to that for the modelled tree (as in Eq5-5 for *E. regnans*):

$$understorey\_effective\_root\_volume = (3.67097/1.76^2)DBH^2 \quad \text{Eq5-15}$$

An alternative method derived from section 5.2.6.3 is to calculate the aboveground biomass (Eq3-4), assume 15% of total biomass is roots, then calculate effective root volume as 66.7% of total root volume (as for the modelled myrtle tree). When using Eq5-15 for understorey trees with DBH <1m in a typical forest stand the root volume appeared to be overestimated compared with expected values for that biome from the literature and therefore the eucalypt formulae Eq5-11 should be used for those trees.

5.4.4 Cross-section

The deviation of the enclosed area from the area of a circle with the same DBH, due to flutes in the buttress and general acircular shape of the stem (the 'acircular area deficit' at 1.3 m) is the gap between the solid line and lower dashed line in Figure 5-12.

The potential enclosed cross-sectional area at 1.3 m rose linearly with DBH^2, though not as steeply as for trees with circular cross-section, due to the acircular area deficit (Figure 5-12):

$$wood\_area = aDBH^2 \quad \text{Eq5-16}$$

where *wood_area* is in m^2 and DBH is in m. (For circular trees a= $\pi/4 \approx 0.7854$.)

The deficit, as a percentage, was twice as variable for trees 2.8≤DBH≤4.5 m – 21(8)% than for larger trees— 21(4)% for trees with 4.5≤DBH ≤7.16 m (Figure 5-13). Saplings have a deficit closer to 0%.

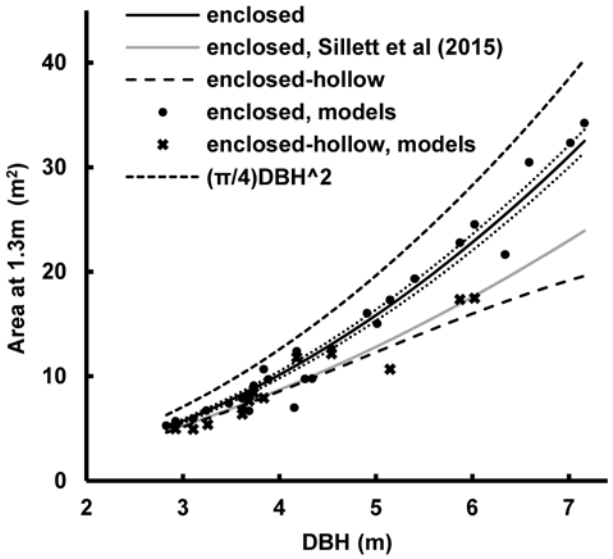


Figure 5-12. Potential wood area at 1.3 m versus DBH.

Solid, black line is regression fit to 3D models, dotted lines are 95%CIs. 'enclosed-hollow'= enclosed minus hollow. Gap between upper black dashed line and black solid line= acircular area deficit.

Hollow area at 1.3 m was given by:

$$hollow_{area} = (aDBH) + b \qquad \text{Eq5-17}$$

where *hollow_area* is ≥0 and in m^2, and DBH is in m. Without the two trees that were solid at 1.3 m (no basal hollow) the hollow area was given by:

$$hollow\_area = (aDBH)^b \qquad \text{Eq5-18}$$

where *hollow_area* is ≥0 and in m^2, and DBH is in m.

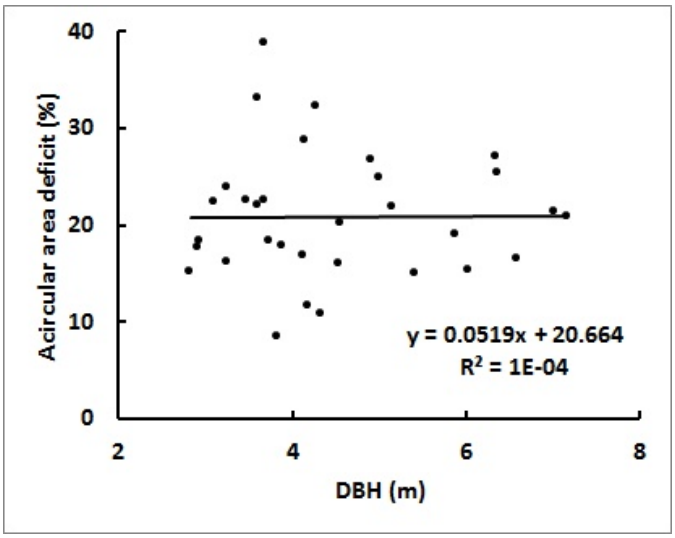


Figure 5-13. Acircular area deficit versus DBH.

The deficit is likely to be more variable for younger and/or smaller mature trees. This variability will influence that of stand-level humus volume.

5.5 Discussion

5.5.1 Methodological implications

The dense point cloud and subsequent 3D surface provided by Photoscan ensured that the models were of sufficiently high spatial and colour resolution to yield valuable tree allometry. This surface detail and the portability of the equipment (camera, light-weight scale bars and small markers) make the methodology described here superior to that available with terrestrial LiDAR (including hand-held, mobile), for such forests and targets. Terrestrial LiDAR equipment with sub-centimetre resolution is prohibitively difficult to transport through mountainous primary forest. Numerous vantage points are required for relatively-accurate surface derivation of tree trunks (Bauwens, 2014; Bauwens et al., 2014). Additional vantage points were required for complex buttress and root-ball environments in the present work, e.g. from above. These vantage points require steady footing for the tripod, which is often unavailable in Tasmanian primary mixed-forests. Hand-held, mobile scanning lasers (Bosse et al., 2012; Hosoi et al., 2013) are suitable for moderately difficult terrain

and intermediate understorey densities but the measurement resolution is in the order of 0.02 m for small trees without epiphytes (Bauwens et al., 2014), which is too imprecise for deriving tree allometrics, and it is likely to worsen in more-complex environments. Mobile LiDAR technology may however be the most suitable for topography and stand-density mapping in support of aerial photography or airborne LiDAR over dense canopies. The 3D models derived here also provide an archive for biological observations in the future (such as stem taper and moss area) in case the trend of demise of remnant large trees continues.

The methodology shown here sidesteps destructive sampling. Ideally for volume calculation of the humus mound and lower trunk, a 3D-model of a freshly logged tree would be made with the humus present, then the humus removed to reveal only the timber and mineral soil, and a second 3D-model made. The difference between the two models would equate to the humus volume plus any buried coarse woody debris (CWD). This would provide more accurate allometrics and therefore carbon accounting but would initiate carbon emissions and would kill the epiphytes.

Many allometrics derived non-destructively miss the trunk hollow area (e.g. Sillett et al., 2015) and the non-circularity of the trunk perimeter. Knowledge of these attributes allows users of allometrics derived from non-destructive sampling to have results as accurate as if deploying destructive sampling, but without the carbon emissions and conservation debt. The methodology used here allows calculation of such attributes, though for some hollow-area assessment, trees were fortuitously found after logging.

The formulae deduced here, for footprint, humus area, humus volume, and area deficit, can be used at the stand-level on inventory data, and when linked with example SOC data for such trees, can be used to provide adjustment to more-routine soil sampling. In typical SOC assessments, SOC is sampled away from large trees and their main coarse roots. The roots encountered in those samples are used to adjust bulk density to give a SOC stock per unit area. The method assumes a uniform root density in between trees that also applies beneath tree trunks and buttresses. The

root density is rarely published and consequently it is necessary to provide an adjustment for the soil bulk density (and SOC) based only on the effective root volume calculated in the present study. Thus, in effect, during calculations using formulae from the present work, the root volume under the trunk and nearby is duplicated by an amount equal to that typically found in between trees. Therefore the method suggested here is conservative for estimating SOC per unit area. The roots of myrtles are more likely to occupy existing hollow roots than are those of eucalypts (as the exterior of myrtle roots appears to be more decay-resistant than the interior Figure 5-14, which was not observed for *E. regnans*)— this also reduces the soil displaced by roots and therefore diminishes the unaccounted-for SOC for a given DBH (though this was part of the present modelling). Some tropical trees exhibit similar growth patterns (Hallé et al., 1978). The SOC stocks adjusted for the forest attributes as shown here can be used to provide more accurate carbon accounts for primary forests and LUC (as in the following chapter).

Figure 5-14. Roots growing inside larger roots reduce effective root volume. The displaced SOC by the roots will be less, but this is taken into account in the modelling presented here.

5.5.2 Example application: buttress hollows and cross-sectional area

The difference between Eq5-17 and Eq5-18 represents a substantial difference in forecast hollow area by including solid trees, though solid trees were exceptions (e.g. Helms, 1945) and those measured were found over 50 years ago. With the prevalence of increased fire and logging in the district meanwhile, hollow-free trees are now less likely to be present. The historical situation is however important for modelling the effects of past emissions on current greenhouse gas concentration. It could be that such solid trees were too exceptional to include in a small sample size, and that a specifically targeted data collection is warranted, during logging of a virgin stand.

Errors arise in measuring annual growth or even NPP of mature primary forests because there is an unmeasured increase in diameter for senescing *E. regnans* trees, as they grow new sapwood, heartwood and roots inside hollow trunks. It was noticed that during advanced senescence of *E. regnans* new sapwood can grow in internal trunk hollows, near the flute folds, and it subtracts from the hollow-area. New heartwood within the tree hollow in *E. regnans* was noticed up to at least 15.4 m aboveground. Similarly myrtles can grow inside their lower trunks. When this is combined with the fluted shape, only a portion of the radial growth will register on a diameter tape (compared with if the tree was circular and hollow-free). Therefore, a measured increase in DBH may not be a reliable indicator of the growth. Additionally, height may not be a reliable indicator of growth as *E. regnans* can increase biomass through epicormic growth after crown loss, especially as neighbouring *E. regnans* continue to succumb to stand self-thinning and wind stress may subsequently reach lower heights.

If applying the formulas derived here to other studies, to trees measured only by their DBH, then the actual area of timber at 1.3 m height is given by Eq5-16 minus Eq5-17 (or Eq5-16 minus Eq5-18 depending on the prevalence of solid trees). It must be noted that the basal hollows generally did not go all the way up the trunk but were roughly conical in shape and followed the silhouette of the buttress taper. This means

that when calculating tree biomass it should not be assumed that the cross-sectional-hollow-area at 1.3 m height is applicable to the whole trunk.

Regarding the cross-sectional area deficit, the results differ from those of Sillett et al. (2015), in that the area deficit in the present work was smaller— my trees had more wood (Figure 5-12). Their trees were smaller (maximum DBH of 5.53 m compared with my 7.16 m), younger, and were mostly from Victoria (Australia). Jaskierniak et al. (2014) showed a slightly larger area deficit of 27% for Victorian *E. regnans* up to DBH =~1.4 m, compared with the 21% in the present work, and they concluded that the measured diameter is more erroneous as tree size increases. Similarly the formulas in Sillett et al. (2015) imply that the percentage deficit increases with DBH. Conversely, Dean and Roxburgh (2006) suggested that the deficit reached 40% for DBH>3.34 m then decreased with DBH. Both Dean and Roxburgh (2006) and Sillett et al. (2015) used tapes to measure flute area, and therefore relied on fewer data and more interpolation. Results in the present work showed that the area deficit approaches a constant value for larger trees of 20.9 (6.6)% and was more variable for smaller DBHs. The data had a minimum DBH of 2.83 m and therefore did not reveal much of the increase for immature trees as did that of Jaskierniak et al. (2014). Combining the present study with that of Jaskierniak et al. (2014) suggests that the area deficit levels off between 1.4 and 2.83 m. In retrospect, the high-end of the data of Jaskierniak et al. (2014) do show a slight trend for reduced deficit. This new information on the physical attributes means that the carbon forecasting model CAR4D, can be updated.

The difference between the remnant forest types in Tasmania and Victoria was noted by Ashton (1981) and linked to the likelihood of surface and humus-fuelled surface fires, and to soil substrate fertility. The Tasmanian forests support humus fires whereas they are absent in present-day Victoria (McCarthy et al., 1999). There is much less humus and fewer myrtles in present-day Victorian *E. regnans* forest (except near the Plenty and Ada Rivers where the 1939 fires did not penetrate and remove myrtles) (Ashton, 1981; Ashton, 1986). Additionally there is an age difference: with fire being less frequent in Tasmania, many of the older stands in the

Styx Valley of Tasmania are currently ~500 year old (Mount, 1964; Wood et al., 2010) whereas the older mature forests in the VCH are generally only ~300 year old (Sillett et al., 2010).

For Western Hemlock (*Tsuga heterophylla*), Julin et al. (1998) found that the area deficit was larger if either the crown was more exposed to wind stress or the footing was unstable. The two Victoria-based studies (Jaskierniak et al., 2014; Sillett et al., 2015) were mostly of younger *E. regnans* than in the present work and one focussed on tall trees, which may be more subject to wind stress [and consequent buttress development]. The higher variability in deficit for the low to mid-sized trees in the present work could represent meta-stable trees (that will die during further stand self-thinning) or emergent dominants. For larger trees, the reduced variation of deficit and convergence on the mid-range (21%) suggests that either the more-wind-stressed trees have either evened out with age, died, or had crown loss (with reduced wind impact thereafter), or that the less-stressed trees (with < 21% deficit) have become the dominants and had an increase in wind impact. The mature, larger *E. regnans* are often shorter and with wide buttresses, and have more open surroundings (Ashton, 1975a). This may be due to the stand being in the latter stages of stand thinning and the loss of gale protection from neighbours has made them lose their crowns but the open space has also allowed more photosynthesis (crown replacement) and they have repeatedly lost and grown new upper branches over a few centuries, and thus increased in girth at the buttress-level. Deeper buttress flutes (greater area deficit) increases the basal humus volume, unless, for example, the wind stress has caused permanent crown depletion, which will also reduce stemflow [of rainwater] and its associated SOC.

Chapter 6 The effect of large trees in primary forests on soil carbon stocks and land-use emissions. Part 2: soil carbon and stand-level effects

6.1 Abstract

Following the arguments in Chapter 5 about typical SOC measurements not accounting for the expected higher SOC concentration close to large trees, SOC and soil bulk density were sampled directly under large eucalypt tree trunks, inside tree trunks, in the humus mounds in the buttress region, and under the humus mounds. SOC was formulated as a function of depth below the solum surface. The cumulative SOC to a depth where SOC was 90% of that in the whole profile, was compared with an earlier measurement of SOC in between trees, away from the trunk and buttress. Formulae developed in Chapter 5 that link SOC, root volume, and buttress shape, to DBH and ground slope were applied to forest stands within 54.4 ha of primary forest. The SOC under large tree trunks was found to be about four times as concentrated as in between trees. Close to the trees, 90% of the total cumulative SOC was estimated to be within ~2.5 m of the solum surface. When the under-trunk SOC was combined with organic soils associated with the buttress region and in nearly decomposed logs, SOC at the unit-area-level increased by ~7(3–12)% relative to the inter-tree SOC alone. For the forest stands measured, that meant an extra 21(10–37) Mg ha^{-1} of SOC. On a global basis, across current temperate, boreal and tropical forests there is an extra 32(1–72) Pg of soil carbon under tree trunks than is presently unaccounted for. This in turn suggests that there are higher emissions from past forest disturbances than have been inferred. The results suggest that the maintenance of large trees will help ensure high levels of forest carbon storage. The protection of medium-sized trees will be necessary to ensure the large trees of the future.

6.2 Introduction

Due to soil rarely being sampled around mature tree trunks, the measured soil bulk density for mature forests is rarely corrected for the concentration of root volume near tree trunks. Yet, the soil and coarse roots previously under tree trunks can be inadvertently sampled after severe disturbance from logging or broad-acre land clearing: a difference which is likely to bias LUC emission calculations. Measurement of SOC and roots closer to and under large, mature tree trunks should thus improve LUC calculations, fully-coupled climate change models, and dynamic global vegetation models (e.g. Ostle et al., 2009).

The present study focusses on *E. regnans*-dominated mixed-forest, which being amongst the most carbon-dense forests (Chapter 1), should provide a relatively high signal-to-noise ratio for variation in SOC (which should therefore be more detectable). Any ΔSOC accompanying the decades of logging of mixed-forest in Tasmania has not been included in national greenhouse gas accounts. The trees could be expected to contribute considerable SOC by central root sloughing during maturation (Ashton, 1975a; Braakhekke et al., 2013). *Eucalyptus regnans* especially, would contribute SOC because of its large and dynamic humus mound, integrating stemflow and hemi-epiphytes and contributing to SOC below the humus mound (Chapter 5). This part of the SOC pool at least, is likely to be affected by LUC in the long-term.

In the present study, potentially important SOC pools associated with large trees were assessed. The macroscopic additions and subtractions to SOC in the buttress region, both above and below ground, underneath and near tree trunks, and the SOC in decomposing logs, were characterised. This was based on tree geometry formulated in Chapter 5, forest-stand-level measurements of DBH and slope, and on soil carbon and CWD measurements. To quantify any difference to total SOC stocks for forest stands more generally, results were expressed as a percentage difference,

the ‘Extra-SOC’, relative to SOC stocks based on sampling the in-between-tree areas only.

6.3 Methods

6.3.1 Study site

Stand-level tree data and SOC data were collected in the Styx Valley; annual rainfall is 1170 mm (1993–2013) with a weak winter maximum (Figure 6-1). The mean daily minimum temperature per month is 2.3 ^{o}C and the mean daily maximum per month is 22.4 ^{o}C.

The soils were Cambisols (IUSS, 2014), in the FAO classification, derived from sedimentary rocks, mostly finer sediments of the Parmeener supergroup. The soils were mountain forest soils, being mostly moderately to strongly leached (podzolic) (Wilde, 1946) to psuedogleyed (Schaetzl and Thompson, 2005). There were three main exceptions. Firstly, soil under the trunks of large trees was dominated by an irregularly shaped bowl of lignomor (decomposed wood, see glossary) ~1 m deep but deeper where coarse roots had penetrated further down (measured up to 2.3 m deep by probing with auger connecting rods). Beneath the lignomor the clay was not affected by either leaching or waterlogging, except if rain had leaked into the top of the tree hollow, in which case there was a weakly bleached horizon under the lignomor, and it was thicker where more water had come from overhead. Where rain entered the main trunk hollows, some roots and more typical-soil-like areas were observed within the lignomor body. Secondly, in two places, where the surface water outside of the trunk in the buttress zone could not flow (due to impediment from the tree’s roots and bedrock) there was a redoximorphic environment, with gleying and a smell characteristic of H_2S. Thirdly, in some locations there were buried albic A horizons, at up to 1.4 m from the solum surface, including their *in situ* charcoal from a forest fire when that horizon was at the top of the solum. These layers had been penetrated and disrupted by tree roots.

Localised deviations near the buttress, from the Cambisol soil type were:

(a) on the downhill side of a large *E. regnans* (DBH= 4.38 m): Sapric Histosol, and

(b) on the uphill side of a medium-sized *E. regnans* (DBH= 3.11 m), where the bedrock was high and relatively impervious and the ground steeply sloping, trapping topographic flow: Histic Stagnosol.

From roadside cuttings it was observed that there were large undulations in the depth to the fresh bedrock surface (e.g. from 0.1–1.7 m depth, within 3 m horizontally), and therefore causing undulations in mineral soil depth, but without noticeable corresponding variation in the biomass of *E. regnans*. This lack of variation in growth maybe because the bedrock was highly fractured and therefore did not impede root growth.

In more waterlogged soils, away from the stand-level plots and further than 40 m from the nearest *E. regnans*, but still in mixed-forests (though of lower biomass) the soils were histic gleysols. The vegetation growing on these soils was *E. obliqua* with implicate rainforest understorey, or myrtle callidendrous rainforest. (For a description of rainforest types see Kirkpatrick and DellaSala (2011)). Elsewhere, if the bedrock was high, and quartz-rich then *E. obliqua* replaced *E. regnans*.

6.3.2 Stand-level tree measurement

Stand-level tree measurements were collected from 10 circular plots of outer radius 60 m, totalling 11.2 ha, randomly located across 54.4 ha of remnant primary mixed-forest, in scheduled logging coupe SX009B. The stand-level data were collected between November 2010 and April 2015. Measurement methodology, including trunk diameter at breast-height (1.3 m) (DBH), and coarse woody debris (CWD) dimensions and decay category (soft, medium or hard) followed the same protocols as in Chapter 3. The site had a callidendrous rainforest understorey. The numbers of

different tree species measured was: *E. regnans* 104, *E. obliqua* 7, *E. delegatensis* 25, myrtle 265, sassafras 839, celery-top pine 19, and five other understorey species 10. Carbon in live-trees was estimated using established formulae (Dean et al., 2003; Dean et al., 2004; Chapter 3), with adjustments for *E. regnans* senescence and site growth potential as detailed below.

Although primary forest, the study area was a vestige left during the industrial high-grading process (Chapter 4) and was therefore not one of the potentially more-C-dense sites in the region. This allowed interpolation rather than extrapolation for biomass and SOC formulas. It may also make the findings more widely applicable, as elsewhere *E. regnans* forests can have high biomass relative to other forest types (Keith et al., 2009; Fedrigo et al., 2014).

The plots were randomly located within the proposed logging coupe SX009B. The furthest distance between plots was 1.3 km. They had a range of topographies and forest types, including uneven-aged stands and seemingly even-aged, mature to mature-senescent. In terms of the major contribution to biomass in the different plots, seven were *E. regnans*-dominated, one was *E. delegatensis*–dominated, one was marginally *E. obliqua* dominated, and one was rainforest-species dominated. All had substantial amounts of *E. regnans*. The geology of the plots was Permo-Carboniferous sediments except for the most southerly one which had Cambrian metamorphic sandstone and mudstone (MRT, 2011). Measurement error in DBH of live trees was estimated to be 5%, and in each dimension of coarse-woody debris (CWD) it was estimated to be 10%. These values were obtained from comparison of multiple measurements of the same objects, and were approximately doubled to acknowledge larger errors in measuring larger trees (Butt et al., 2013).

Formulae for carbon in live *E. regnans* trees (Dean et al., 2003; Dean et al., 2004; Chapter 3), include adjustments for senescence and localised growth potential. Parameters in those formulae were adapted to the stands measured in the present work. The mid-year of senescence was set at 550 years and the rate of senescence

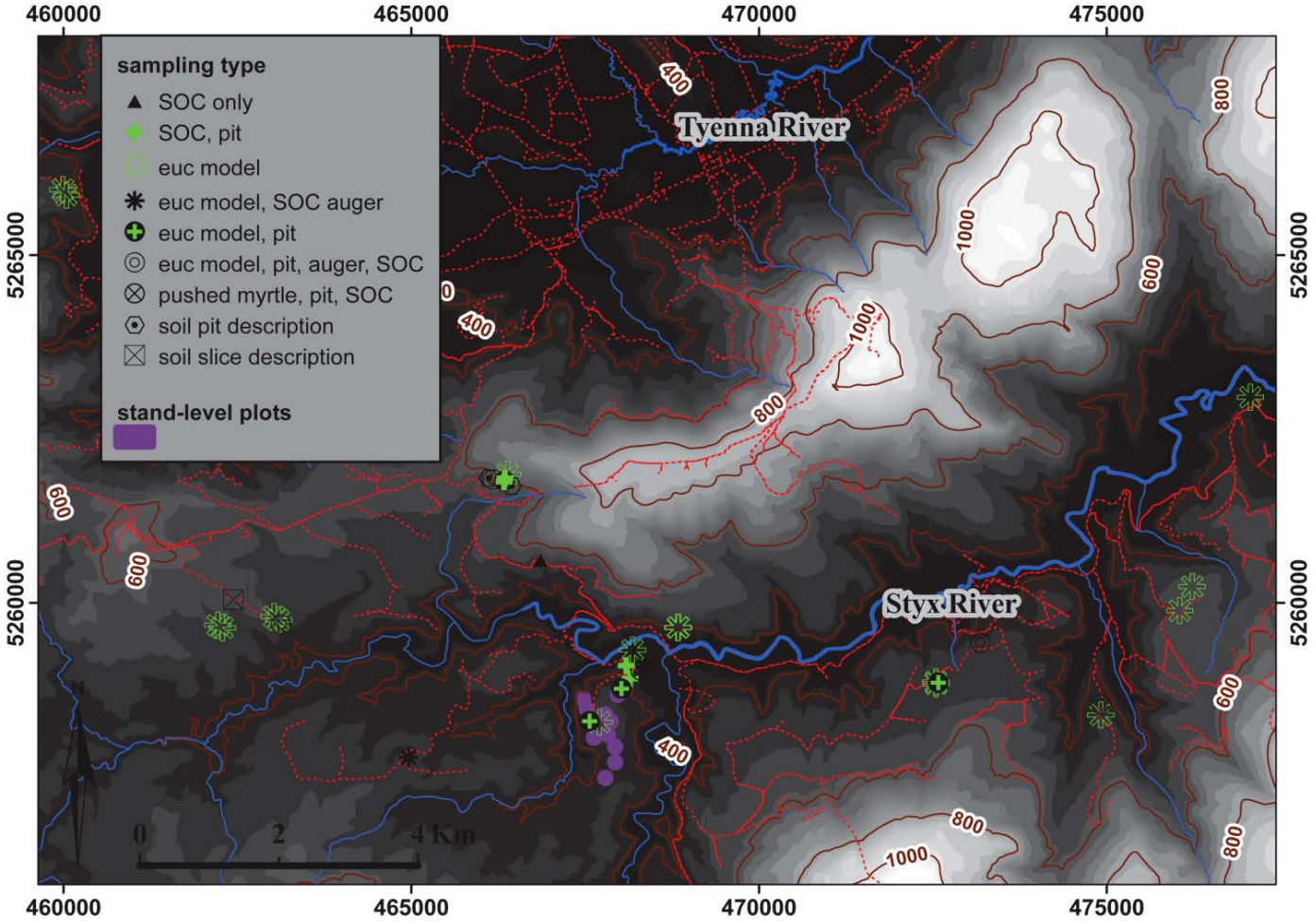


Figure 6-1. Study area showing sample types in Styx Valley and Tyena Valley. 'euc model'= 3D model of *in-situ* eucalypt buttress, 'SOC'= soil organic carbon sampling for carbon assay and bulk density, blue lines= rivers and named creeks, red lines= roads. Non-eucalypt 3D-models not shown. Datum GDA 1994, projection MGA, zone 55, metres.

at -10 (Dean et al., 2003 Table 3.1, Eq.8), and the productivity multiplier for DBH was 1.880 (Figure 6-2). These parameters simulated the biomass peaking at 400 years, and gave a DBH of 5.3 m at 500 years. They were selected to match the age of large mature *E. regnans* in the region being ~500 years (Mount, 1964; Wood et al., 2010), and the maximum DBH in the plots which was 5.20 m. Additionally, with these parameters, aboveground C maximised for a *E. regnans* tree when DBH= 4.943 m, at 62.4 Mg, which compared well with that of 58.7 Mg from formulas in Sillett et al. (2015). For trees used in 3D-modelling the maximum DBH observed was 7.16 m but such larger trees were on more productive sites than in the ten stand-level plots. Using these parameters, whole-tree C, including senescence, was modelled as a function of DBH using Eureqa and applied to the stand-level data:

$$Eregnans\_C = 3.394142DBH^2\{1/[1 + \exp(5.081129DBH - 27.68206)]\}$$

Eq6-1

where *Eregnans_C* is in Mg and *DBH* is in m. The multiplier for understorey biomass was 0.5, following (Dean et al., 2004; Chapter 3).

The root volume of snags at the stand-level was calculated using the formula used for *E. regnans* (Eq5-11) and assigned a fractional presence based on the CWD decay classification (hard=1, medium= $^2/_3$, soft= $^1/_3$). The volume of soil displaced by large roots for each tree in the stand-level data was tallied to adjust SOC per unit-area given by (Dietrich, 2012).

6.3.3 Soil sampling and analysis

When determining SOC stocks at the unit-area-level there are several ways to sample, measure and tally, for the whole soil profile and they give different results. Rarely is the whole soil profile measured even though it is known to be the most accurate method to get SOC per unit area. Instead, either a formula is fitted to the data measured to a specific depth and extrapolated downwards, or the measured data are stated as representing the entire SOC stock, i.e. equivalent to no SOC existing further down. The latter approach can create

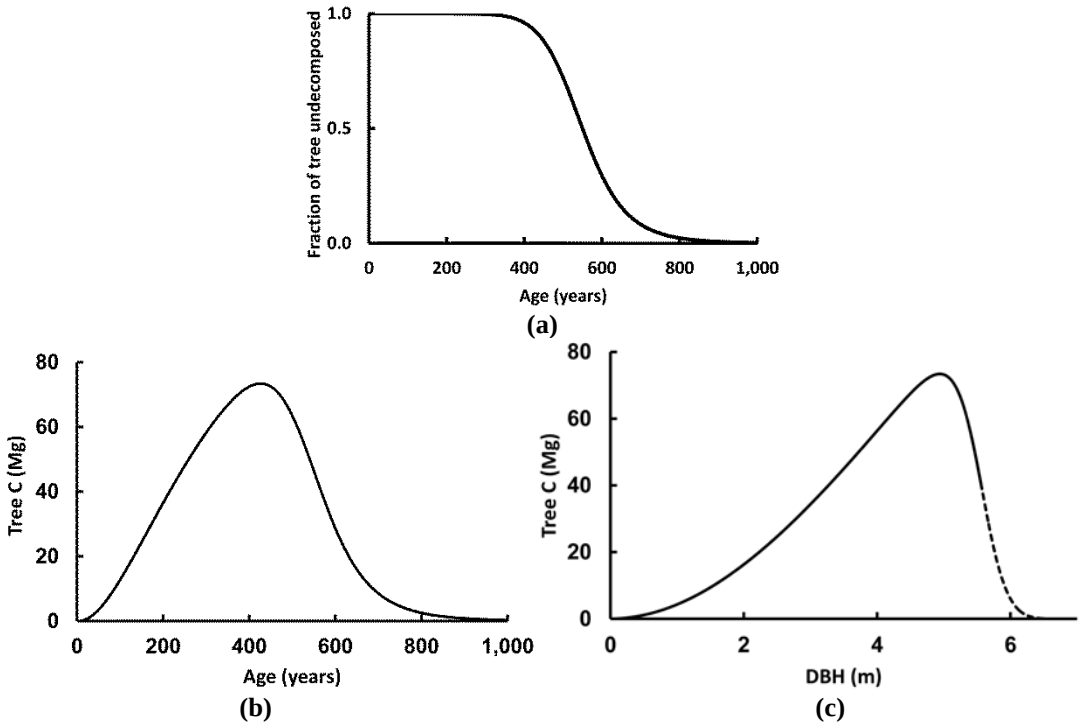


Figure 6-2. Adaptation of *E. regnans* formulae in Dean et al. (2003) and Dean and Roxburgh (2006) for SX009B. **(a)** Fraction of tree that is undecomposed, as a function of age (i.e. inverse of fraction senesced), **(b)** whole tree carbon versus age, derived by multiplying '(a)' by un-senesced tree carbon vs. age, and **(c)** whole tree carbon versus DBH, derived from '(b)' by inverting the equation for age as a function of DBH, note high DBH tail is dashed to indicate some trees of this advanced age may have lost girth at 1.3 m. Note that these adaptations were designed for a depauperate mixed-forest stand in coupe SX0009B in the Styx Valley, and are unsuitable for higher-productivity, less-severely burnt stands.

misconceptions when measuring SOC change (ΔSOC) with LUC. A version of the latter approach when calculating ΔSOC is to measure a constant mass of mineral soil, but that method is applicable where the soil profile and nature of the change are well defined and the amount of change is known to be minor compared with the original SOC stock. SOC is generally older deeper down (Paul, 2016), which implies that it generally changes more slowly there. Therefore measuring deeper layers can also create misconceptions regarding the effects of LUC if there is insufficient time between measurements (e.g. when removing

trees) or insufficient time for several generations of trees to grow, mature and decompose (e.g. when adding trees to a long-cleared landscape or changing the dominant tree species), thereby emulating the new ecosystem-type for which one intends to make a claim.

Measuring the SOC in the whole soil profile as a single unit can be done either by analysing all the soil dug from a pit, or by extracting a representative number of entire-profile cores then measuring their carbon content. These methods were impractical for the present study as the necessary machinery could not be transported over rugged, mountainous terrain whilst leaving the forest intact, and therefore they were not used here.

Prior to collection of soil samples for carbon analysis several pits were dug beside or underneath trees that had been measured and detailed by photogrammetry, in order to determine the general nature of the soil horizons. In the field, soil pH for the surface (0–0.05 m) and subsurface soils (0.05–0.55 m) were determined with a field testing kit (Raupach and Tucker, 1959), accurate to 0.5 pH units.

Soil samples were collected from inside trees, under trunks and in the outside of the buttress region, from 1.6 m below the solum surface to 2 m above it (Figures 6-3, 6-4, 6-6). Above the solum surface in the buttress region, the soil near the spurs was sampled, and inside the tree above the solum surface lignomor was most commonly sampled. Soils were sampled by augering in pits and augering from the surface without pits, with an Eijkelkamp© soil corer (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands) for later elemental and bulk density analysis. The procedure used with the Eijkelkamp© auger is described in Roxburgh et al. (2006a). Where pits were dug, such as inside walk-in-trees or on tree buttresses then the Eijkelkamp© hand-held bulk-density head was used for sampling exposed soil.

The number of profile cores was determined by practicalities. For example the trunk hollows in some trees were narrow and tall and allowed only one core whereas others were wide and allowed both a pit and multiple cores at the same level (Figure 6-4).

Soil samples were air dried at 20º C. Stones >0.01 m and roots >0.0005 m were removed from the samples, weighed, and their volumes measured. Samples destined

for elemental analysis were ground in a mortar and pestle followed by grinding to a fine powder using a ball mill (MM200 Mixer Mill, Retsch GmbH, Haan, Germany). Wood samples (for spectral comparison with soil and for carbon assay) were finely split with a chisel, then ground in a mortar and pestle, and finely ground in a domestic blender.

Removed stones were ground separately and assayed for C, as if they were soil. Soil volumes, bulk densities and carbon densities were adjusted for roots. Stone C was re-included as SOC because significant amounts of organic carbon can be dissolved into stones or adsorbed on their surface, with significant contribution to total SOC (Harrison et al., 2003; Zabowski et al., 2011).

The pH of 18 representative distinct soil samples was determined after drying and grinding, by two-point calibration of a Hannah pH Meter on a gently stirred mixture of 1:5 soil:distilled water.

The analysis for total nitrogen, carbon and hydrogen in 93 soil and seven wood samples used a Thermo Finnigan EA 1112 Series Flash Elemental Analyser, which was operated by Thomas Rodemann at the University of Tasmania.

Pedogenic carbonates may precipitate as an illuvial horizon as high up as at -3(0.4) m from the soil surface under 1170 mm $year^{-1}$ rainfall (Royer, 1999) (the rainfall of the study site). Effective rainfall was lower under tree trunks, potentially making the precipitation depth sufficiently shallow and within the sampling depths of the present work. This necessitated checking soil carbon for inorganic carbon as opposed to assuming it was all organic carbon.

Inorganic C analysis was performed by John Gouzos of CSIRO, Adelaide and his method followed Sherrod et al. (2002) and Rayment and Lyons (2011a). Inorganic C was determined by reacting the sample with acid in a sealed container and measuring the pressure increase. Sufficient finely ground sample to contain no more than 0.8 g $CaCO_3$ equivalent was weighed into a 250 mL glass bottle, a tube containing 8 mL 3*M* HCl and 3% ferrous chloride added and the bottle sealed. The contents were

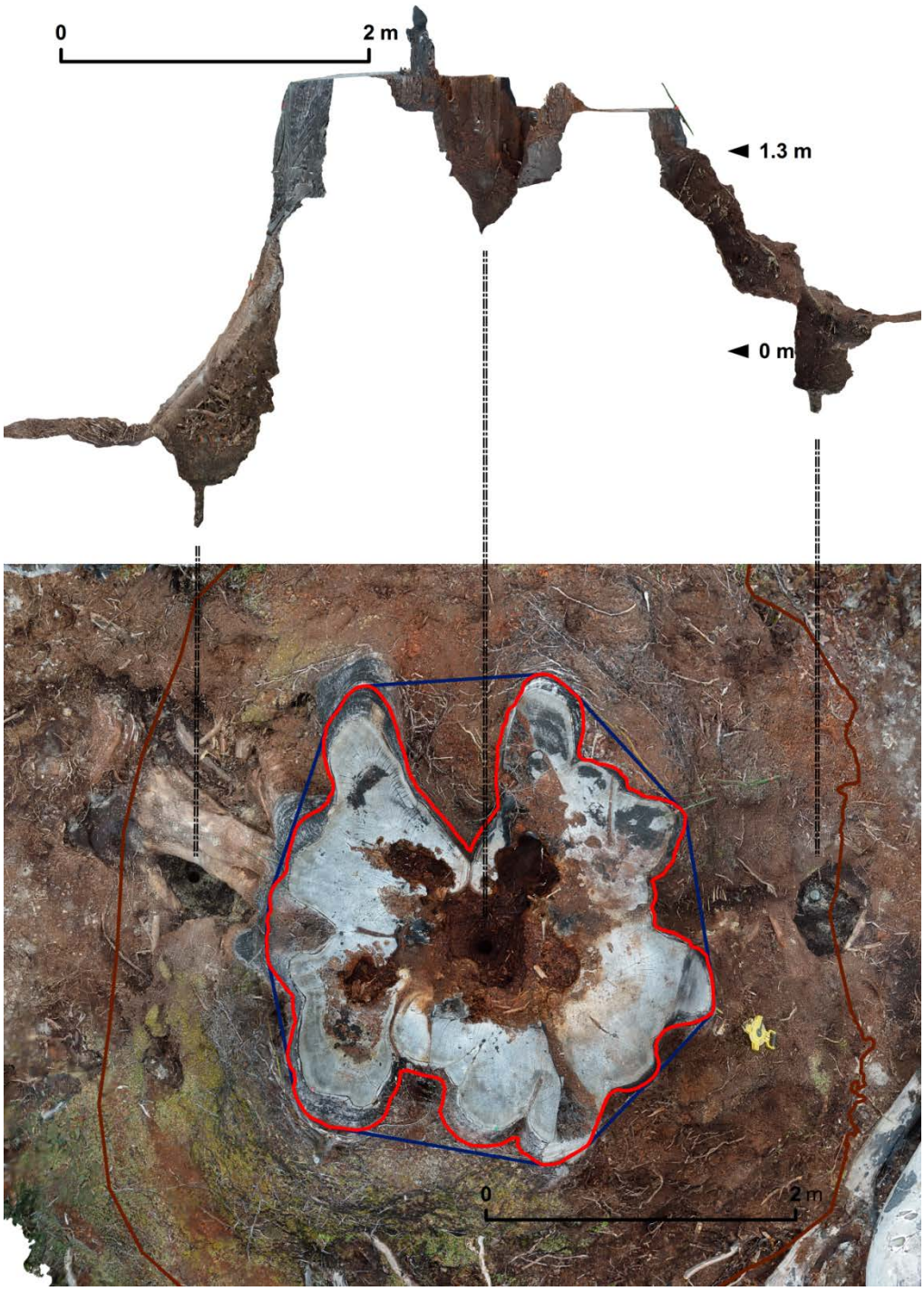


Figure 6-3. Example soil core locations. Cross-section and top view of *E. regnans* stump after logging, DBH= 3.11 m (same tree as in Figures 5-3 and 6-4.d), showing upper parts of augered core locations: two cores in the under-humus region and one in the under-trunk and inside tree (lignomor extended aboveground). Entire holes not visible as camera could not view hole bottom from multiple

angles. Height shown on cross-section refers to tree measurement, whereas soil augering used local height basis.

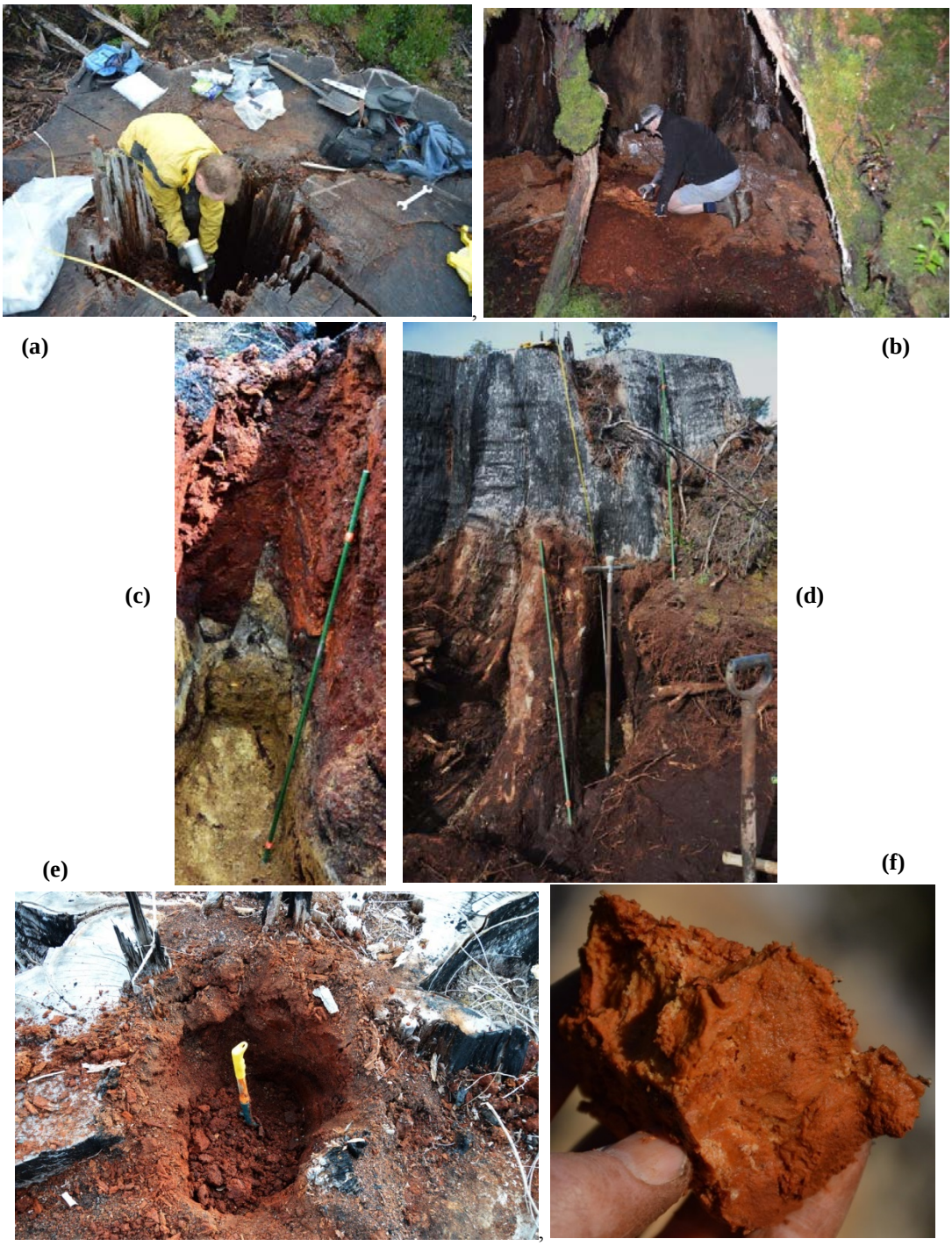


Figure 6-4. Varieties of soil sampling. **(a)** auger only (photo: R. Doyle, 2013), same tree in Figure 5-5, **(b)** pit and auger, inside 'Chapel Tree', centre: Dr Doyle (Figure 5-1.c & Figure 6-6.a), **(c)** pit below burnt and fallen *E. regnans* (lignomor above yellow clay). **(d)** augering in the under-humus region (same tree in Figures 5-3 and 6-3), **(e)** pit and auger, same tree as in '(d)', **(f)** lignomor from inside trunk of a pushed-over myrtle, showing absence of original timber structure.

mixed intermittently during a 1 hour period and the pressure in the bottle measured by piercing the septum with a needle attached to a pressure transducer.

The highest inorganic C proportion, was for one stone at -0.785 m. The amount of inorganic C was only 2(2)% (0.27–5.8%) of total soil carbon, with the highest values for stones, which were infrequent in the cores. Therefore, the portion of soil carbon that was not derived from forest biomass was negligible, and the low pH of the samples did not justify further assaying for inorganic carbon. (Low pH means dissolving and leaching of inorganic carbonates.)

Data from under-trunks and under-humus were analysed separately. SOC density data for aboveground soil inside-trees were not differentiable and were merged, similarly for those in the humus mound. SOC data away from trunks and the large tree footprint were from Dietrich (2012) who measured forest floor C and the SOC from -0.45 to 0 m from the solum surface.

Other SOC data were available for *E. regnans* forests in the State of Victoria Polglase et al. (1994). Both that data set and that of Dietrich (2012) were sampled away from tree trunks and large roots and were only down to -0.3 and -0.45 m for Victoria and Tasmania respectively.

6.3.4 SOC data processing

Four methods for estimating cumulative SOC per unit-area as a function of depth were tested and are explained below. When deriving formulae, parameters were refined by nonlinear regression using LABFit (da Silva and da Silva, 2015) as described in Chapter 5.

Method 1 (M1): Where SOC density can be expressed as a mathematical function of depth, the cumulative carbon per unit area can be calculated by analytical integration

of the function from 0 m to a particular depth (e.g. Jobággy and Jackson, 2002) or to minus infinity to give the cumulative carbon for the [whole] mineral soil (Ogawa et al., 1961). Infinity is an approximation to include a fractured C horizon, its heterogeneity and some dissolved organic carbon (DOC) The simple exponential decay function for SOC density:

$$SOC\_density = a[\exp(bz)] \quad \text{Eq6-2}$$

where *z* is the distance from the solum surface (in metres, negative below 0 m) and *SOC_density* is in $Kg \cdot m^{-3}$, appears to have been used first by Ogawa et al. (1961). Its integral:

$$cumulative\_SOC = (10a/b)[1 - \exp(bz)] \quad \text{Eq6-3}$$

where *cumulative_SOC* is in $Mg \cdot ha^{-1}$, gives the cumulative SOC with depth (Ogawa et al., 1961; Roxburgh et al., 2006a; Mishra et al., 2009). This is an approximation as contrasting soil horizons are likely to cause oscillations in slope of cumulative SOC/depth. Thus the change in SOC density with depth is unlikely to be a simple exponential function over the whole soil profile and may even not be monotonic, due to *inter alia* different properties of horizons (Waksman, 1938; Kempen et al., 2011) but here it was used as a first approximation.

Method 2 (M2): Where the formula for SOC density as a function of depth is too complex to integrate analytically, the density values can be used to approximate the unit area stock for small depth intervals (e.g. 0.001 m), i.e. per hectare, and these tallied over depth (i.e. numerically integrated) to give the total stock per hectare. The method of numerical integration used in this work was 'trapezoidal'. The errors it produced were checked and it was found that for 0.001 m intervals over the soil profile, the value of cumulative SOC down to ~50 m depth (assuming permeability to organic material) when compared with analytical integration, gave an error of only $1x10^{-5}$ %, which was considered negligible. However the data were not of sufficient precision or uniformity to warrant complex equations and therefore M2 was not further used.

Method3 (M3): The SOC density can be measured for specific broad layers of soil that may correspond to soil from visually different 'horizons', or to depth differentiation as determined by available technology, e.g. a hand auger. Measurements within each layer are averaged, then the depth-span of each layer is applied to give a unit area of measurement for specific layers (but only those measured), e.g. giving SOC stock in Mg ha^{-1} for the 0-to-0.1 m layer or the 0.1-to-0.3 m layer. Then the stocks for consecutive layers are added together. This is the method commonly used to report results down to a specific depth. It was used for *E. regnans*-dominated forests in Polglase et al. (1994) and in Dietrich (2012). Sometimes it is reported or interpreted as if the whole profile was measured, leading to misconceptions about SOC stock.

Method 4 (M4): I introduce a method suggested here for the first time [as far as I know] that is more appropriate to SOC density measurements covering a broad range of depths not associated with predetermined layers, and that is suitable for merging data with spatially diverse SOC densities. Data were collected at a wide range of depths because of the irregular nature of the sampling environment, *viz*. large, cut stumps with lignomor in the trunk hollow above the level of the outside mineral soil surface; cavernous, live hollow trees; fallen trees; and deep humus between buttress spurs (Figure 6-4). Depth is divided into 0.001 m intervals. SOC density data from within the 0.05 m high sampling ring were separated into these intervals to represent the vertical position within the ring (Figure 6-5). Data from spatially different locations, but representing the same soil type (e.g. within a trunk hollow), were merged. If more than one data point was present for a given 0.001 m interval then those data were averaged. Intervals missing any measurement were assigned the average value from the nearest measurements above and below it. The interval data were then tallied to give cumulative SOC with depth, from the solum surface down to the bottom of the sampling ring for the deepest measurement. This gives a thousand data points per metre depth, which would lead to a statistically unrepresentative, small standard deviation. Therefore, for formulation of a function representing cumulative SOC, a subset of SOC cumulative data, one point for the centre of each sampling-ring position, was extracted. These were then graphed and

formulated as a simple exponential function of depth, from which cumulative SOC for the profile is readily calculated:

$$cumulative\_SOC = g[1 - \exp(bz)]\,, \qquad \text{Eq6-4}$$

which can be differentiated to yield SOC density:

$$SOC\_density = gb[\exp(bz)]/10 \qquad \text{Eq6-5}$$

where units are as in Eq6-2 and Eq6-3 for variables with the same names.

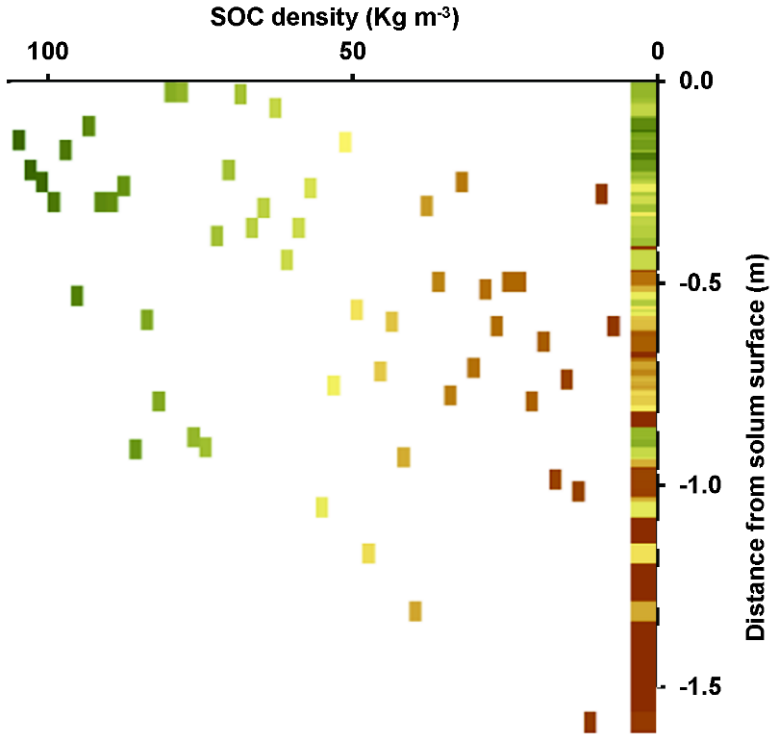


Figure 6-5. Schematic illustration of method M4, using inside-tree data. Concentration data are position-coded and colour-coded: green and left= high-SOC grading linearly to brown and right= low-SOC. Each coloured box represents a sampling point with a sampling-ring, with the ring height of 0.05 m having 50x0.001 m intervals.

M4 was more accurate than M3 for the data measured here. M3 underestimated the cumulative SOC (Figure 6-8), and consequently it was not used for the final analysis. Inside the centre of tree trunks, the lignomor was continuous across z= 0 m height [from the solum surface level determined outside the trunk]. Thus data above, and below 0 m but within 0.025 m (half the sampling ring height) provided a more

reliable value for SOC density at 0 m (termed ‘C0’), and this value was used in the method entitled ‘M4-fixed C0’, which was the method used for stand-level SOC calculations. C0 was the average carbon density of the lignomor. Equations for under-trunk SOC were trialled with and without the value at z= 0 m being held constant at C0.

Nonlinear regression played a larger part in determining the parameters for M1 than for M4 due to the greater spread of data for M1, and the uneven data point distribution [with depth] for M1 may have biased its derivation by regression. The 95% confidence intervals for M4 were relatively narrow because of the dependency of deeper SOC data on shallower data in M4, and therefore the variability of the deeper data is occluded in M4.

In the final analysis SOC was calculated by M1 and M4 only. M4 was considered to be the most accurate and M1 is that most commonly used in the literature. Equations for M1 and M4 were fitted to three main regions: under-trunk, under-humus (under the humus mound), and in between trees, the latter based on data from Dietrich (2012).

For SOC concentration as a function of depth, and for cumulative SOC, several simple equations were tried, as suggested by the program Eureqa (Schmidt and Lipson, 2009) and that were considered stable in nonlinear regression analyses (Ratkowsky, 1990). These possibilities were tested in case the data were not best suited to a simple exponential decline with depth. For under-trunk SOC density, two equations that were more complex than simple exponential decay (requiring method M2 to yield cumulative SOC) gave the same cumulative SOC but had larger error margins and there were insufficient data to reveal a strong horizon effects corresponding to their curvature. Therefore they were disregarded.

6.3.5 Stand-level effects

The study of SOC in between trees by Dietrich (2012) included five mixed-forest plots in the Styx, Tyenna and Florentine valleys and one of his plots was centred

within my stand-level site. His work included 31 samples from mixed-forest. The forest-types in that study were similar to those in the present work: primary mixed-forest, mostly *E. regnans*-dominated but Dietrich (2012) had higher amounts of other dominant species: 20% *E. delegatensis*-dominated, 20% *E. obliqua*-dominated, and 20% myrtle-dominated. The data of Dietrich (2012) for mixed-forest were used as the background benchmark to which the effects studied in the present work were compared mathematically.

The soil below the cross-section of live trees at 1.3 m was assigned the SOC range of concentrations 'under-trunk' (e.g. Figure 6-6.a) and it was surrounded by the under-humus soil. The error in assigning this diameter to the under-trunk soil could possibly be as high as 30%, as judged from digging inside hollow trees and below fallen trees, but it errs on the side of caution (in terms of SOC tallying) as the high-SOC under-trunk soil was often observed to go beyond the DBH distance for larger trees. This area of soil was tallied for all live trees and snags by methods M1 and M4. Snags were considered to contribute a portion, pro-rata, according to their decomposition category (1, $^{2}/_{3}$ and $^{1}/_{3}$ for hard, medium and soft respectively) towards altered, under-trunk SOC. It is possible that snags may have had higher SOC than indicated by their decomposition category due to all their roots undergoing decomposition. However that had not been determined and instead it was calculated as if, for example, a 'soft' category of dead tree was two thirds of the way to being entirely absent from accounts.

The equation for humus volume as a function of a tree's footprint (Eq5-9) was applied to each live eucalypt tree with DBH≥1 m and summed at the stand-level. This was converted to SOC ha^{-1} using the SOC density for the humus mound, 91(13) Kg m^{-3} (section 5.4.2). The SOC beneath the buttress humus mound was determined from the equation for humus area (Eq5-8) and the cumulative under-humus SOC by M1 and M4. Understorey trees were not noted to have humus mounds and as many were hemi-epiphytes on the eucalypts any rainforest-species humus mounds (if they did exist) may well have been shared ones and therefore already counted. Snags were considered to have no humus mounds. A few dead eucalypts had vestigial humus

mounds but generally the humus volume was minimal as litterfall decreased with advanced senescence and epiphytic trees died with decrease in stemflow.

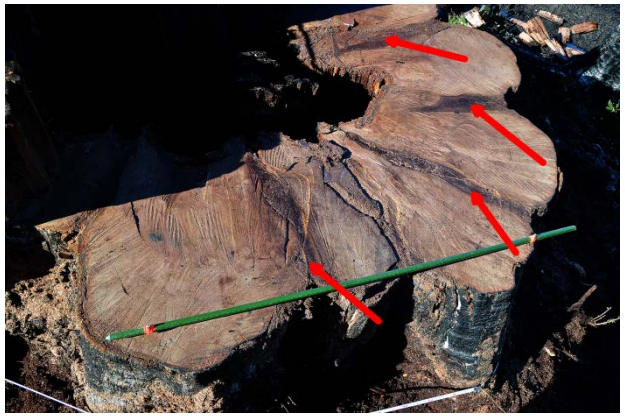
In some trees in the non-decomposed heartwood-to-sapwood region of the buttress, between the central hollow and the bark, the sides of the neighbouring spurs had grown towards each other and compacted the flute, i.e. they had filled the inter-spur void. Over decades or centuries the bark inclusions trapped within the wood had decomposed to form a peaty soil (e.g. Figure 6-6.b–d) and in some places it included coarse and fine living roots of epiphytes (indicating it was regarded as soil by other trees). This soil was considered of negligible mass per hectare and not tallied into the stand-level accounts.

Similarly, aboveground lignomor ($z>0$ m) in the trunk hollows, was not included in the stand-level tally. An estimate showed that 26 large trees per hectare containing lignomor up to 1.3 m in their trunk hollows would be required to increase the SOC per unit area by 1%. However only a couple such trees simultaneously were noted in each logged coupe. This infrequency is probably because the trees can survive several hundred years with hollows but the lignomor may decompose during that period, especially if there is a basal fissure.

Another probably minor contributor to total SOC was a form of lignomor that I have called ‘cunnite’ (Appendix II), after its similarity in appearance to black coal, its similar carbon density (804 Kg m^{-3}) to anthracite, and because it was only found in myrtle trees (*N. cunninghamii* (Hook. f.) Øerst.). Cunnite was a dark, hard, dehydrated (mummified) form of lignomor found in most senescing myrtle trees, where the fresh, wet product was light coloured and with equivalent conchoidal fracture (Figure 6-4.f), which I called ‘myrtle cheese’. In pure form its volume at the stand-level was probably a few orders of magnitude below that of the other soil types assayed. Therefore it was excluded from regression equations for cumulative SOC and from the stand-level tally.

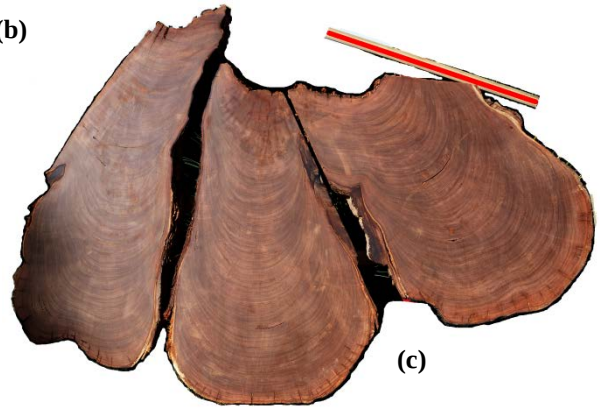
Stand-level SOC data were tallied from all the 10 plots. The SOC was firstly calculated as if all soil was similar to that measured for the region by

(a)



(d)

(b)



(c)

Figure 6-6. **(a)** Soil pit inside 'Chapel Tree' a live hollow *E. regnans*. (Figures 5-1.c, 6-4.b), showing 0.94 m of lignomor derived from roots, above clay. **(b)**, **(c)** and **(d)**: bark inclusions in buttress from previous flutes— SOC not included in analysis.

(b) and **(c)**: extreme example, *E. obliqua* in coupe TN050G, DBH= 2.92 m. (c) is slice taken from right-hand side of (b) with timber in fastened in original position— after partial drying the peat fell out, red scale bar is 0.5 m. **(d)** *E. regnans* in coupe SX019I. Note shoes in lower left for scale.

(Dietrich, 2012) for in-between trees. The SOC was next calculated by replacing the areas under-trunk and under-humus with the SOC values calculated in the present work, and adding the humus mound SOC. The relative difference between the two methods was the 'Extra-SOC', expressed as a percentage of the first method.

6.3.5.1 SOC in well-decomposed logs, 'soft-logs'

Chemical and physical attributes of CWD in common Canadian species (Strukelj et al., 2013) were compared with those for the under-trunk lignomor found in the present work (there are no similar data for Australian cool-temperate forests). The attributes of the lignomor: bulk density, SOC density and C/N ratio (Table 6-1), were within one standard deviation of those of 'decay class-V' of the Canadian CWD (186.20(21.2), 95.27(13.1) and 197.00(97.5) respectively (standard deviations from their reported data)). The colour of decay class-V in northern-USA forests is red-brown to dark-brown and has virtually no visible wood structure (McFee and Stone, 1966; Maser et al., 1979). This matches with the characteristics of *E. regnans* lignomor (red-brown) and myrtle lignomor (dark-brown, though light-brown to orange-yellow in one tree), although fresh lignomor was yellow to orange (though red-brown to dark-brown or black once the fungal hyphae had died).

The 'decay class-IV' of Strukelj et al. (2013) had too high a C/N ratio to match lignomor observed here. The C/N ratio was the elemental composition attribute of CWD with the highest standard deviation (close to 50%) in both the present study and in Strukelj et al. (2013), this being due to the steep decline with decomposition, the patchy nature of decomposition stage within a cross-section of timber and possibly also due to variability between species of live wood. Different tree species sampled in the present study varied significantly in C/N (by up to five times).

A three-category classification system was used for CWD (hard, medium and soft) in the present work, with the 'soft' category corresponding to 'decay class V' and some 'decay class IV' of Strukelj et al. (2013) and Grove et al. (2011). Grove et al. (2011 Fig. 3) found ~30–35% of decay classes IV plus V to be non-rotten wood in *E.*

obliqua forest. Thus it seems conservative to suggest that half of the soft-log mass (in the present work) is equivalent to 'decay class V' and therefore is SOC rather than wood.

The similarity indicates that logs of an equivalent decomposition stage in the present study area could be considered as soil rather than wood, i.e. half of the 'soft-log' category CWD corresponds to 'decay class V' plus some 'decay class IV', also known as 'soil wood' (Jurgensen et al., 1997). It is likely that there is also SOC in 'medium' class CWD but the proportion is too indeterminate to include here and in order to remain conservative with respect to SOC, it was excluded.

At the stand-level, half of the soft-log C was added to the SOC tally.

6.4 Results

6.4.1 General soil characteristics

The pH values for lignomor (except myrtle coal), and non-lignomor soil in the vicinity of lignomor, were 'ultra-acidic' (terminology of Rayment and Lyons, 2011b), whereas all other substances were only 'extremely acidic'. The value for non-lignomor solum under the trunk 3.40(0.24) was similar to the value of 3.6(0.6) for Tasmanian rainforest soil (di Folco and Kirkpatrick, 2013). The lignomor was more acidic, with pH= 2.73(0.12). Stones in the vicinity of lignomor had pH= 3.95 (Table 6-1).

Variation in C0 (the lignomor SOC density under-trunks at z= 0 m) was primarily due to variation in bulk density, with its standard deviation (SD) being ~25%, which in turn was mostly due to varying water content, with a SD of ~16%, whereas the SD of C wt% was only 6% (Table 6-1). The variation in water content was most likely due to lignomor in trunk hollows having been exposed to rain by logging but not to sunlight for several years (low bulk density) whereas that in the upper levels of unlogged hollow trees was slowly dehydrating and compacting (high bulk density).

6.4.2 Carbon density and cumulative SOC

Bulk density vs. soil carbon (C) wt% showed a hyperbolic relationship (Figure 6-7), as noted elsewhere (e.g. Huntington et al., 1989; Périeé and Ouimet, 2008; Hiederer and Köchy, 2011), but for the full range of C densities (from stones to lignomor), up to C 50 wt%. It was not necessary to form a log-log relationship for bulk density versus C wt% as in, for example, Huntington et al. (1989):

$$bulk\_density = a + b\sqrt{Cwt\%} - c\ln(Cwt\%) \qquad \text{Eq6-6}$$

where *bulk_density* is in kg m^{-3} and *Cwt%* is the percentage carbon of soil by weight, *a*= 1408.2(64.6) kg m^{-3}, P(t)< 0.005, *b*= 72.959(46.135) kg m^{-3}, P(t)= 0.117; *c*= 431.41(70.67) kg m^{-3}, P(t)< 0.005; df= 88, adjR^2= 0.78, (probability) *P*< 0.005.

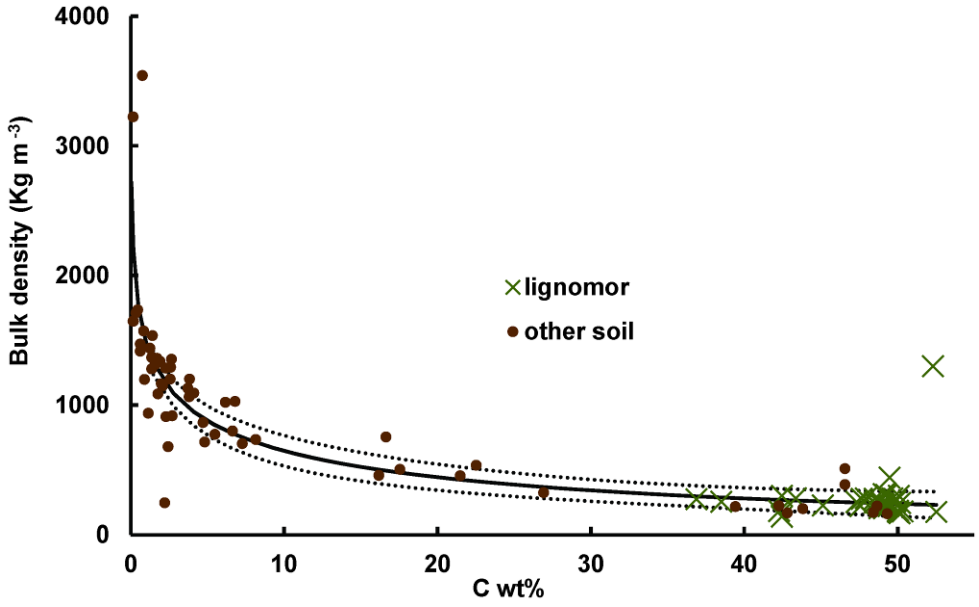


Figure 6-7. Bulk density vs C wt%, formula in Supporting Results. Dotted lines are 95% CIs. The outlier at the right-top is cunnite, which was not included in the regression.

Table 6-1. Average attributes for specific soil types. Only from cores without contaminates of other soil types. Standard deviation in brackets.

Soil type		Num	Distance from solum surface (m)		Bulk density (Kg m^{-3})	C density (Kg m^{-3})	C wt%	C/N	H_2O wt%	pH
			min	max						
lignomor without myrtle coal/cheese		29	-1.07	2	238(59)	115(29)	48(3)	128(70)	70(11)	2.73(0.1)
lignomor		30	-1.07	2	274(201)	138(129)	48(3)	128(69)	68(15)	
non-lignomor mineral soil		37	-1.6	-.115	1090(403)	31(25)	6(11)	24(12)	27(15)	3.40(0.24)
mudstone in deepest cores		2	-0.611	-0.785	3380(225)	16(16)	0.47(0.43)	17(11)	48(8)	3.95
inside-tree	aboveground	9	0	2	270(82)	131(43)	48(2)	187(88)	66(18)	
	belowground	51	-1.6	-0.025	749(718)	70(44)	25(22)	64(50)	50(24)	
outside-tree	aboveground	8	0.05	1.36	279(125)	91(13)	37(11)	34(7)	57(14)	
	belowground	19	-1.40	-0.05	1070(403)	32(20)	7(15)	22(7)	24(8)	
palaeosol A horizon with charcoal		2	-1.4	-1.4	1644	1.4(2.0)	0.19(0.02)	3.8(0.6)	17	3.97
charcoal		2	-0.3	-0.3	444(89)	221(22)	50(5)	408(56)	-	
myrtle coal (cunnite)		1	0.3	0.3	1296.7	804.03	52.30	130.75	16	3.62
myrtle cheese		1	0	2	310.6	152.51	49.10	233.81	69	

The two best fitting solutions for representing SOC down the soil profile were M1, (Eq6-2) and M4, (Eq6-4) (Figure 6-8, Table 6-2). Method M2, numerical integration of M1, was unnecessary; and method M3, averaging SOC density for macroscopic layers, was inapplicable to the soil structure.

Table 6-2. Solutions to Eq6-4 (M4). Standard deviation in brackets.

	g (Mg ha^{-1})	P(t)	*b* (m^{-1})	P(t)	adjR^2	Df	*P*
under-humus	516.312 (23.688)	<0.0005	0.840134 (0.058057)	<0.0005	0.997	18	<5x10^{-8}
under-trunk	1202.23 (33.69)	<0.0005	0.992529 (0.041934)	<0.0005	0.995	56	<5x10^{-8}

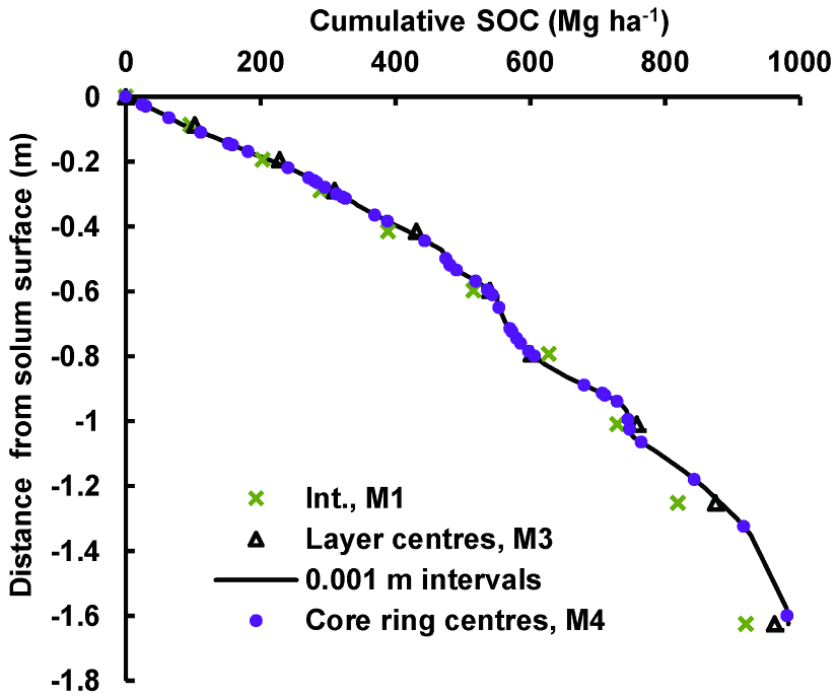


Figure 6-8. Comparison of methods' results for under-trunk cumulative SOC. 'Int.'= integrated.

Relatively high variability in under-trunk C density arose from the contrast between lignomor and the mineral soil, and the spatial variability of previous roots. The SOC densities for the M1 and differentiated-M4 differed only by a maximum of 7% down the profile. The effect on M4 of fixing SOC density at 0 m to the lignomor average (C0) was

negligible because it only had a localised effect, not multiplicative down the profile, as would have been the case with M1. For both under-trunk and under-humus soil, M4 was chosen as most representative for cumulative SOC.

The SOC that could be practically measured in routine future experiments was considered as being represented by 90% of the cumulative total SOC as calculated here from equations. For all equations 90% of total SOC was within 2.5 m of the surface (Table 6-3). Results for the cumulative SOC (Table 6-3, Figure 6-9) showed substantially more SOC closer to trees than in-between-trees: 4x for under-trunk and 1.7x for under-humus.

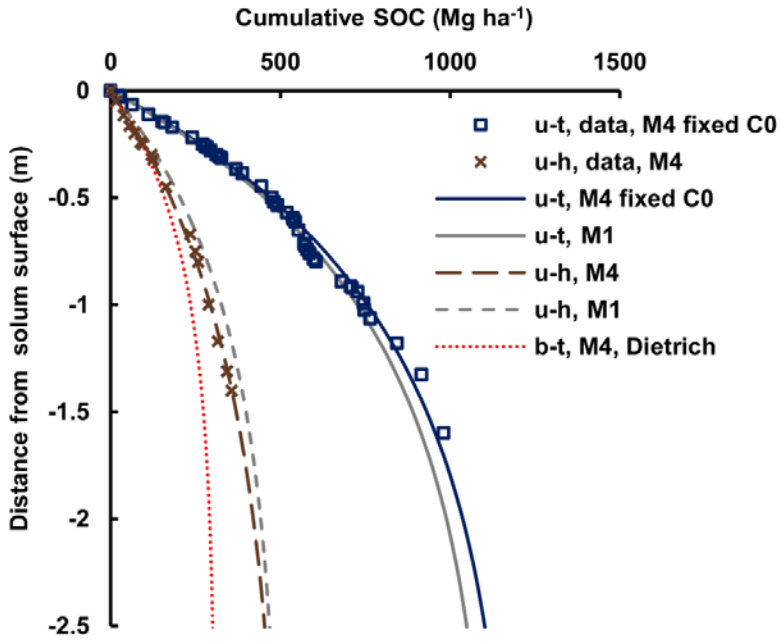


Figure 6-9. Effect on SOC of proximity to large trees, aspatially. Cumulative SOC, data and regression equations, u-t= under-trunk, u-h= under-humus-area, b-t= in-between-trees.

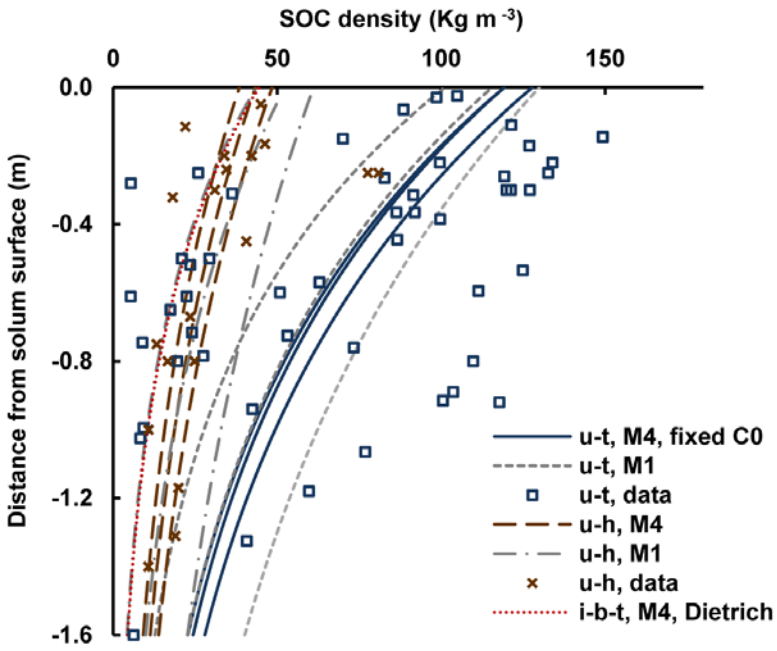


Figure 6-10. SOC density location comparisons, for M1 and M4. 'Diff.'= differentiated, 'u-t'= under-trunk, 'u-h'= under-humus area, 'b-t'= in-between-trees. Each regression curve for u-t and u-h, is a triplet, with the two outer curves being 1 standard deviation of regression parameters.

Table 6-3. Cumulative SOC, and depth at 90% of total. Numbers in brackets are 95% confidence intervals.

Method	Restraint	Cumulative SOC (Mg ha^{-1})	Depth at 90% of total (m)
		under-trunk	
M1	none	1138 (781–1768)	-2.27
M4	C0 at 0 m	1202 (1137–1277)	-2.32
M4	none	1223 (1152–1306)	-2.44
		under-humus	
M1	none	505 (300–986)	-2.22
M4	none	516 (473–572)	-2.73
		in-between-trees, derived from Dietrich (2012)	
M1	none	264 (144–605)	-1.26
M4	none	309 (295–324)	-1.59

The cumulative SOC down to only 0.3 m under-humus was: M1 135 (81-263) Mg ha^{-1} and M4 115 (106-128) Mg ha^{-1}. This was marginally different to that for SOC in-between-trees in the region (Dietrich, 2012): 106(22) Mg ha^{-1}, but not significantly so. The in-between-tree value for a similar forest type in the Australian State of Victoria (Fedrigo et al., 2014) was higher, at 169 (156-182) Mg ha^{-1} but it is possible that the Victorian *E. regnans* mixed-forest has higher SOC corresponding to its higher biomass (Fedrigo et al., 2014).

6.4.3 Stand-level carbon

Carbon in live biomass (including roots) was 448(134) Mg ha^{-1}, 33(17)% of which, i.e. 156(82) Mg ha^{-1}, was from understorey trees. Carbon in logs was 64(50) Mg ha^{-1}, and C in snags was 62(61) Mg ha^{-1}.

Carbon in live biomass was correlated with the basal area of trees with DBH≥ 1 m (Figure 6-11):

$$C\_in\_live\_biomass = (a\ large\_BA) + b \qquad \text{Eq6-7}$$

where *C_inlive_biomass* is in Mg ha^{-1} and *large_BA* is in m^2 ha^{-1}, a= 5.11034 Mg m^{-2}, SD= 0.63117, P(t)< 0.005; b= 117.474 Mg ha^{-1}, SD= 45.827, P(t)= 0.03; df= 8, $P < 1x10^{-6}$, R^2= 0.89, adjR^2= 0.88, $P< 5x10^{-5}$. The intercept on the ordinate axis, i.e. parameter b, represents the average C in live biomass for trees with DBH < 1m. Trees with DBH≥ 1 m comprised 69(38)% of total C in live biomass. Eq6-7 shows the importance of large trees with regards to C stock and that equation can be used for a rough approximation (i.e. by only measuring trees with DBH≥ 1 m) of stand-level biomass in mixed-forests on similar sites and with similar fire history (thus allowing collection of more field data with the same person power).

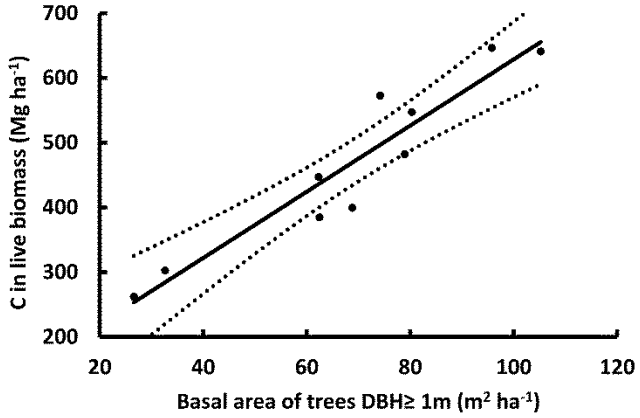


Figure 6-11. At the stand-level, the strongest indicator of C in live biomass was basal area of large trees. Dotted lines are the 95.4% confidence intervals.

Basal area was 118(62) m^2 ha^{-1}, of which 52% comprised eucalypts with DBH≥ 1 m. The corresponding under-trunk Extra-SOC was 2.8(1.6–4.0)% (numbers in brackets represent confidence intervals), 74% of which was for trees with DBH≥ 1 m. The under-humus SOC of eucalypts with DBH≥ 1m raised the SOC per unit-area by 0.35(0.20–0.60)%.

The overall effect of including near-tree SOC in mixed-forests was 7(3–12)% more SOC (for M4) than is usually reported for such forests (Table 6-4). The equivalent amount from M1 is higher: 8(5–12)%. M4 was considered more reliable than M1 due to the higher likelihood of bias during regression for M1.

There was 1,273(45) m^3 ha^{-1} of live-tree coarse root volume close to trunks, with 87% of that coming from trees with DBH≥1 m; and 46(34) m^3 ha^{-1} from snags. This coarse root volume from live trees and snags reduced SOC stocks by only ~0.63%.

Table 6-4. Effect on cumulative SOC (to 90% of total) from 'Extra-SOC', compared with in-between-trees (Dietrich, 2012). Numbers in brackets are 95.4% confidence intervals.

Method	In-between-trees (Mg ha^{-1}), derived from Dietrich (2012)	New total SOC (Mg ha^{-1})	Extra-SOC (%)
	All 10 plots		
M1	264	284	7.9(4.7–12)
M4	309	330	6.9(3.2–12)
	Seven *E. regnans*-dominated plots		
M1	264	287	8.9(5.4–14)
M4	309	332	7.6(3.6–13)
	Two most C-dense plots		
M1	264	296	12.4(7.4–20)
M4	309	341	10.6(5.0–18)

The single largest contributor (40%) to the Extra-SOC at the stand-level was from under-trunk C of living trees, this being from decomposed roots (Figure 6-4.c, Figure 6-6.a, Figure 6-12). The majority (67%) of that came from eucalypts, and of that 66% from eucalypts with DBH≥ 1 m. The under-trunk SOC for living trees and snags together contributed the majority (55%) of the Extra-SOC.

For eucalypts with DBH≥ 1 m the SOC stock in the humus mound was only 3.5(1.1) Mg ha^{-1}, and it raised the SOC per unit-area by 1(0.70–1.2)%.

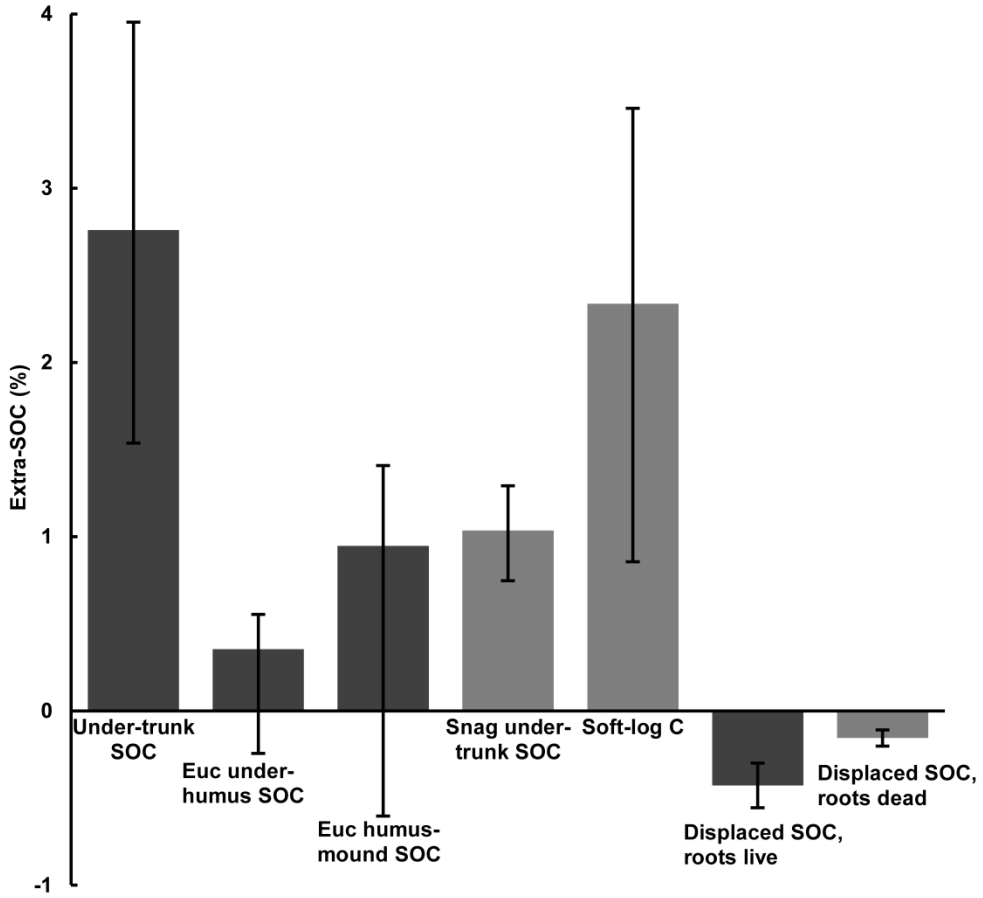


Figure 6-12. Overall effect of including near-tree and soft-log SOC on unit-area 90% of SOC stocks. Dark-grey— living trees, light-grey— dead trees.

Soft-log C contributed ~34% of the total Extra-SOC (Figure 6-12). It must be appreciated, however, that soft-log C is normally counted as CWD C and is therefore already part of typical whole-of-forest estimates. The amount of soft-log volume at the stand-level was 129(128) m^3 ha^{-1}, half of which was attributed to lignomor: 64(64) m^3 ha^{-1}, which is less than the volume of decay-class-V CWD found for the same forest type by Dietrich (2012), namely 82 m^3 ha^{-1}. This comparison suggests that the amount of soft-log C was not overestimated in the present work.

The effects of snag under-trunk SOC was approximately the same as that of the buttress humus (Figure 6-12), 15% and 14% contributions to Extra-SOC respectively. The smallest single contribution, 5% of the Extra-SOC, is under-humus SOC. The SOC displaced by roots (live and dead) contributes -0.58% to the Extra-SOC.

Results in Table 6-4 are presented for groupings of the plots: all 10 plots, *E. regnans*-dominated plots only, and the two most-carbon-dense *E .regnans* plots. Sites from the work of (Dietrich, 2012) could not be differentiated into forests types as their SOC values were all within one standard error of each other and the means of the *E. regnans* and *E. delegatensis* forest types were equal to within three significant figures.

For the *E. regnans*-dominated stands only, and especially for the two more-even-aged of those, the change in Extra-SOC from 7(3–12)% to 11(5–18)% was contributed mostly by large, dead trees. The Extra-SOC for these stands should perhaps be compared with the inter-tree SOC from that specific forest type rather than with the general inter-tree SOC of Dietrich (2012) but the standard deviations in that study did not allow differentiation of forest types. A more comprehensive set of observations might show that for different mixed-forest types the percentage Extra-SOC might stay the same although the absolute Extra-SOC (in Mg ha^{-1}) changes.

Regardless, the contribution from snags to the increase in under-trunk SOC and the SOC displaced by roots, were both doubled on going to the two more-even-aged *E. regnans*-dominated stands. The contribution from soft-logs increased by ~70% and the under-trunk SOC for live trees with DBH≥1 m increased by ~45%. Thus, overall, the contributions that increase SOC in the absence of fire for *E. regnans* mixed-forest are from the large trees (dead and alive) and from soft-logs. This trend is equally applicable to sites logged earlier in the region, noting that the region has been subject to high-grading (Chapter 4). This has implications for LUC accounting on more productive land globally.

At the stand-level, the attributes most strongly correlated to Extra-SOC were the basal area of trees with DBH ≥1 m (R^2= 0.53, *P*< 0.02), (and consequently biomass was also strongly correlated, Eq6-7), humus volume (R^2= 0.58, *P*< 0.02), and the projected area of logs with diameter ≥ 1 m (R^2= 0.77, *P*< 0.001). The first two correlations are probably a reflection of how the Extra-SOC was formulated— with SOC close to trees being functions of DBH, and the assumption that trees with DBH<1 m only had half as much Extra-SOC.

A question that arises is under what circumstances can root volume (the negative influence on SOC per unit-area) outweigh the major positive influences on Extra-SOC? The two largest contributions to Extra-SOC shown in the stand-level data were SOC under-trunks and soft-log volume. The Extra-SOC under-trunks with DBH≥1 m was positively correlated with obstructive root volume of living trees: adjR^2= 0.97, *P*< $2x10^{-7}$; and this relationship is dependent mostly on the obstructive root volume of eucalypts with DBH≥1 m: R^2= 0.85, *P*< $2x10^{-4}$. Therefore as the root volume from large eucalypts increases and displaces SOC so too does the lignomor from dead roots, which adds to SOC. The second largest contribution to Extra-SOC, soft-log volume, was not significantly correlated with root volume. Therefore the negative impact of larger root volume cannot override the positive effects on Extra-SOC, at least not for mature forests.

Stand-level formulas with only large trees as the dependent variables allow more field data collection with the same person-power in future studies on similar sites. At the stand-level, humus-area was a linear function of basal area of eucalypts with DBH≥ 1 m:

$$humus\_area = (a\ euc\_large\_BA) + b \qquad \text{Eq6-8}$$

where *humus_area* is in $m^2\ ha^{-1}$ and *euc_large_BA* is in $m^2\ ha^{-1}$, *a*= 0.713576, SD= 0.052271, P(t)<0.005; *b*= 8.66461, SD=3.480955, P(t)=0.04; df=8, *P*< $1x10^{-6}$,

adjR^2=0.95. Stand-level humus-volume was a linear function of basal area of eucalypts with DBH≥ 1 m:

$$humus\_volume = (a\ euc\_large\_BA) + b \qquad \text{Eq6-9}$$

where *humus_volume* is in $m^3\ ha^{-1}$ and *euc_large_BA* is in $m^2\ ha^{-1}$, *a*= 0.343093, SD= 0.086612, P(t)=0.004; *b*= 16.7382, SD=5.780, P(t)=0.2; df=8, *P*< 0.005, adjR^2=0.62.

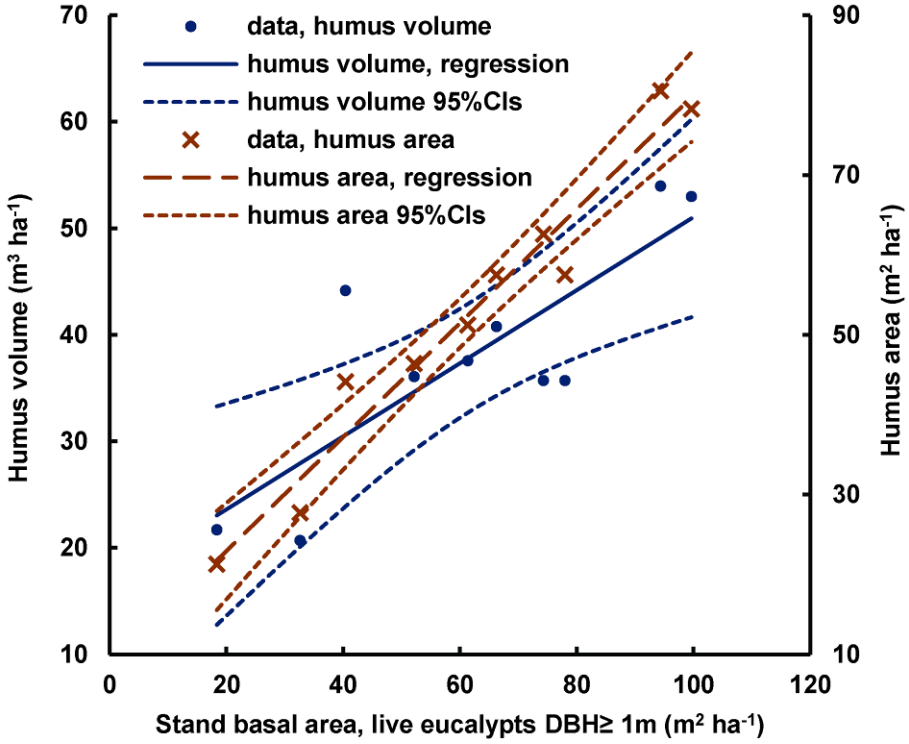


Figure 6-13. At the stand-level, humus-area and humus-volumes were linear functions of basal area of eucalypts with DBH≥ 1 m.

Note that the above equations for humus-area and humus-volume cannot be used directly to calculate humus amounts. However, their parameters when calculated by standardised major axis linear regression in program SMATR (Warton et al., 2006) can be used thus:

in Eq6-8, humus area: *a*=0.7284, 95%CIs 0.6718–0.8589; *b*= 7.728, 95%CIs: -0.241–15.698

in Eq6-9, humus volume: *a*=0.4126, 95%CIs: 0.2668–0.662; *b*= 11.89, 95%CIs: -1.39–37.924.

6.5 Discussion

6.5.1 Implications and applicability

The consistent bias of 7% would obscure measurable SOC change resulting from logging, or broad-acre clearing, for several decades in situations where the upper solum, lignomor and humus mound were mixed during, which in turn could contribute to poorer policies for climate change mitigation.

The findings here are relevant to the management of the tropics and boreal zones where the largest tracts of primary forests remain (Wirth et al., 2009a). They also are relevant to temperate forests such as the Douglas Fir (*Pseudotsuga menziesii*)-dominated

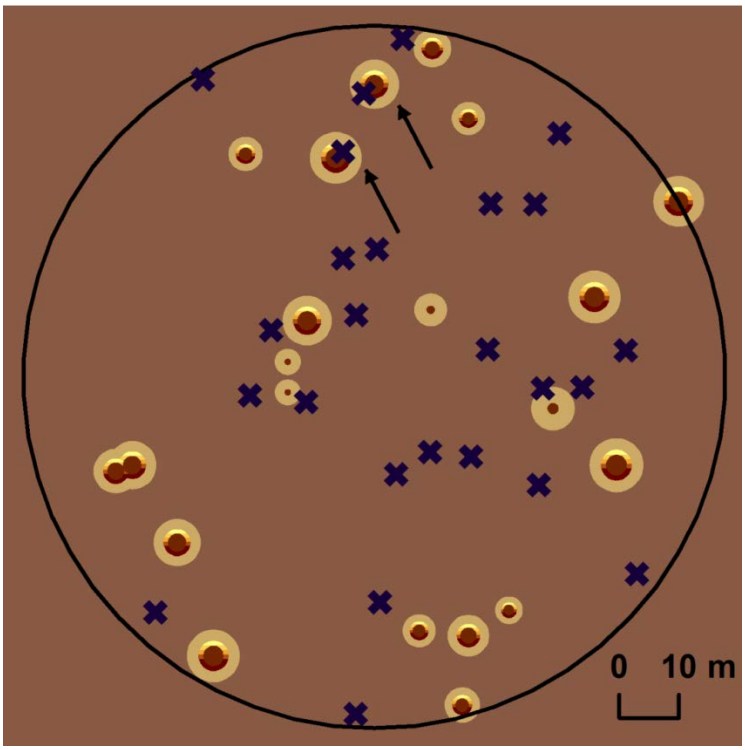


Figure 6-14. Schematics of how heterogeneity in SOC is usually missed at the stand-level prior to logging. Spatial distribution of trees with DBH ≥1 m in forest plot #8. Circles drawn to scale. SOC levels: centred circles= under-trunk, varying shade middle rings=under-humus and humus mound, outer lighter

rings= obstructive root zone (1.5x footprint), background= in-between-trees. Crosses= random soil sampling. Where points are selectively excluded (or relocated) due to inaccessibility— pointed to by arrows— sampling would miss more-dense areas and obstructive root zones prior to logging, but less likely to miss after logging, especially with upper solum redistribution.

rainforests and Coastal Redwoods (*Sequoia sempervirens*) of N. America, which can accumulate humus mounds in the absence of fire (Dominik DellaSalla and Steve Sillett, personal communication, 2015). Many coniferous forests lack high humus mounds around the base of dominant trees, except possibly where fire is absent for over a century, but then the humus may be more-evenly distributed. For example, prior to the logging-fires in the Canaan Valley (WV, USA), a climax red spruce (*Picea rubens* Sarg.)-dominated forest in boreal climate conditions, had a widespread, permanent, deep humus-moss layer (Allard and Leonard, 1952) (Figures 6-15.d, .e).

Higher stocks of SOC can also be reported from deeper soil sampling and deeper soils (Jobággy and Jackson, 2002; James et al., 2014). In the present study the organic C density in bedrock (mudstone) was non-negligible: about half that of the non-lignomor mineral soil (Table 6-1). The mudstone was approximately half water and thus susceptible to organic C addition from dissolved organic carbon. Additionally the upper bedrock was fragmented, thereby possibly distributing C to deeper rocks. The effect of not measuring the entire profile on the calculated SOC per unit area is not known due to the contrasting influences of rocks containing SOC and possible transport of carbon in groundwater. However it is most likely a minor contribution to the penultimate error margins as it is likely to affect both under-tree and between-tree equally. It would be pragmatic to provide some limit on the depth to which SOC is tallied, and for where the soil data were collected, the Styx Valley, I suggest 4 m to include >90% of the SOC in the solum and weathered parent material and some SOC permeating bedrock (or at least to 2.5 m to represent 90% of the SOC). Mobile groundwater, found in several places in the study region, would redistribute C, and confound C modelling. The soil depth below the solum surface for the remnant *E. regnans* forests in the Styx Valley, Tasmania and

Figure 6-15. Loss of high SOC concentration. **(a)** Logging debris piled on eucalypt stump to ensure humus, rainforest species and CWD are burnt during CBS. **(b)** Root lignomor lost to at least length of bar, 1.5 m (top in red circle), due to CBS, from a live hollow *E. regnans* at edge of logging coupe (photo: Y. Bar-Ness, 2013). **(c)** Loss of SOC under-trunk and in decomposed root channels in a live *E. marginata* (jarrah) Western Australia due to wildfire. **(d)** and (e) Roots bared after forest floor humus layer burnt during logging almost a century ago, Dolly Sods, West Virginia, USA.

Victoria (the Victorian Central Highlands, VCH) differs greatly, with the VCH being much deeper in places (5–20 m, Ashton, 1975a; Polglase et al., 1994) and the portion of the Styx Valley sampled in this study, limited to 2–4 m (being principally of glacial or peri-glacial origin). Also, parts of the VCH have much higher biomass than measured here (Keith et al., 2009).This could mean a different SOC profile with depth for the VCH.

Logs may cover up to 12–20 % of the ground area in mature and oldgrowth Douglas Fir-Western Hemlock (*Pseudotsuga–Tsuga*) (Harmon et al., 1986). Some have suggested a slow integration of well decomposed CWD into the forest floor matrix (Strukelj et al., 2013). Work in the present study suggests that a portion of it is already SOC, in the form of lignomor. For consistency in carbon accounting, if soft-log C cannot be considered as SOC then neither can the decomposing roots, which would need reassignment as CWD. Subsequently that volume could not be considered as soil and therefore the bulk density and SOC per unit-area would need to be decreased. Indeed, some authors refer to a decay class system for coarse roots, although the most decomposed category (possibly lignomor) was too malleable to be measured by the techniques employed for the less decomposed classes (Mobley et al., 2013). Thus it was inadvertently considered as soil, though typically not measured during soil sampling. For the sake of consistency this approach would also require redefining *in situ* decomposing trunk and branches to the CWD pool rather than the current practice of counting them as aboveground biomass. In forests elsewhere, SOC can also be on branch forks and adventitious growths (e.g. Sillett and Bailey, 2003; Sillett and Van Pelt, 2007; Wooley et al., 2008). The mineral soil and forest floor are not necessarily boundaries for soil carbon accounting but the consequences of the more diverse SOC repositories is undetermined. Attributing soft-log C to the soil pool rather than the CWD pool may be controversial, however the re-assignment will help rationalise carbon dynamics modelling, easing hiatuses therein. If some portion of ‘decay class-IV’ is also lignomor then more C should be re-assigned to the SOC pool.

The SOC associated with large trees, such as in the humus mounds, decomposing coarse roots and decomposing logs, although most often unmeasured, is inherent (though undifferentiated) in many carbon dynamics models, where SOC is derived from tree carbon but treated as homogenous in 2D or 3D (i.e. bulked). Tuning such models requires adjustment of bulked-SOC half-lives to match observed SOC stock (i.e. calibrating model parameters, Hararuk et al. (2014)), and therefore they are no more accurate than the SOC sampling. In turn such modelling is part of fully-coupled climate change models. Another complexity is that DOC is often inadvertently incorporated as part of the SOC, as only the most complex SOC dynamics models have a separate pool for DOC (Tipping et al., 2012).

The biomass of the understorey is commonly less than that of the eucalypts in mixed-forest (e.g. 30(13)% in Chapter 3 and 33(17)% in this chapter) so their decomposed roots should be the minor contributor to Extra-SOC compared with that from dead eucalypt roots, unless they die more frequently than eucalypts or their dead roots have a longer half-life.

6.5.2 Limits to applicability

There are two scenarios in which under-trunk SOC is unlikely to contribute as strongly as it does in Tasmanian *E. regnans* forests. Firstly, fire increases basal fissures (Adkins, 2006) and thereby increases the probability of C emission from inside-tree lignomor, with higher fluxes as the frequency or duration of forest-floor fires increases. Depletion of inside-tree lignomor was observed in many *E. regnans* after intense logging burns , and observed in most Jarrah forests in Western Australia, which have higher wildfire fire frequency than *E. regnans* forests. Secondly, in some forests the trees may be older than the half-life of the central under-trunk lignomor, which may thus have long-since decomposed (e.g. *Sequoiadendron giganteum*).

For both under-trunk and under-humus SOC profiles, the cumulative SOC calculation method M4 consistently gave higher values than M3, and slightly deeper depths to reach 90% of the total. The small intervals in M4 approximate a spline fit to the cumulative data with a smooth curve, whereas in M3 the data points lie at the midpoint on longer line segments that more-roughly approximate a curve. As shown by Wendt and Hauser (2013) the former types of data fit (M4) is more accurate than the latter (M3), which will underestimate the cumulative SOC. This then leaves the question as to why M4 indicates a higher cumulative SOC than integrated M1, (e.g. 1223 compared with 1145 for under-trunk (7% higher), and 515 compared with 506 for under-humus (2% higher)). The alternative question is does M1 underestimate the cumulative SOC? The only distinguishable reason for the difference is that M1 was more reliant on nonlinear regression for derivation of the equation parameters and there were fewer data points deeper down, which could bias its derivation.

If a lack of spatial heterogeneity in SOC was detected empirically, it would indicate that the present findings were inapplicable. Schöning et al. (2006) detected no spatial variation in SOC in younger forest with an inter-tree distance of 7 m. However, they may not have sampled close to trees or there may have been a high degree of soil disturbance prior to the most-recent tree germination. The two most mature-only plots of *E. regnans* had 15.63 trees ha^{-1}, which gives an average inter-tree distance of 25.3 m, or 17.9 m for tightest circle packing, filling ~76% of the space. Both spacings allow SOC heterogeneity.

6.5.3 Anthropogenic effects

The eucalypt under-trunk and inside-hollow lignomor was ubiquitous but was depleted to varying extents after fires (Figure 6-15). When undisturbed it has the capacity to alter the half-life of SOC per unit-area. It will also influence the spatial heterogeneity of SOC for one or two successive generations of forest, and could explain the legacy of SOC heterogeneity (Wilson et al., 1997; Harmon, 2001). The spatial heterogeneity from humus and stemflow is absent from secondary forests

(Oyarzún et al., 2011), which could reduce soil variability and in turn threaten ecosystem multifunctionality and sustainability, and thereby reduce climate change resilience (Thompson et al., 2009; Havlicek and Mitchell, 2014; Wagg et al., 2014). The burning of lignomor is an emission that must be considered in forest management. The maintenance of large trees, with their localised high concentrations of SOC, will help ensure high levels of forest carbon storage. The protection of medium-sized trees will be necessary to ensure the large trees of the future.

A reduction in the biomass and associated SOC of large trees in tall open-forests in Tasmania is forecast under climate change (Bowman et al., 2014; Chapter 2). Large trees are also set to decline in tropical forests that are already water-limited (Slik et al., 2013). More generally, larger trees become less common as the areas subjected to forestry, anthropogenic fire, agriculture and mining, increase; also with climate change (Galbraith, 1939; Rankin, 1947; Ferguson, 1948; MacFie, 1999; Laurance et al., 2000; Herrmann, 2006; Lindenmayer et al., 2012; Hansen et al., 2013; Jacob et al., 2013; Mackey et al., 2015; McIntyre et al., 2015).

Initially, large tree decline will increase SOC, through a sudden increase in lignomor, but in the long term as that lignomor decomposes it will not be replenished with new input of similar magnitude (as the large trees will not be replaced with more of their ilk). Therefore, in the long-term, temporal-average SOC will be lower.

The positive effect of Extra-SOC with increased basal area of eucalypts with DBH≥ 1 m means that young stands are poor performers. There is little scope for mitigating the carbon emissions associated with logging primary forests, except for longer harvest cycles and increased wood-product recycling (Chapter 4). If the harvest cycle length is considerably shorter than the time taken for root decomposition, then the root carbon may accumulate over multiple cycles. This scenario however may eventually decrease tree productivity rates and decrease other carbon pools.

With 13,545 ha of *E. regnans* primary mixed-forest remaining in 1976 (ANM, 1979) in the Styx and Florentine Valleys the estimate of a 6.9% underestimation for SOC implies an underestimation of 258 (130–458) Gg of SOC in pre-logging carbon (if

assuming the in-between-tree value of 278 Mg ha^{-1} derived from Dietrich (2012), being 90% of total SOC). This is a conservative estimate as 4,000 ha of mixed-forest had already been converted by 1959 (ANM, 1959). The long-term emission of SOC if logging cycles continue, and assuming a long-term (millennial) decrease in SOC of ~50% (Chapter 4), is 2.0 (1.9–2.1) Tg. The contribution to State-wide carbon stocks from *E. regnans* may have also been put at risk by re-seeding with other eucalypt species, of lower maximum biomass, due to scarcity of *E. regnans* seed (Forestry Commission, 1978), but the other species may be more resilient to the fire regime of the future climate.

Decomposition of both below-ground CWD and SOC has been forecast to increase with climate change (Kolchugina and Vinson, 1995; Kirschbaum, 2004), contributing to the positive feedback of climate change (Raich and Schlesinger, 1992). The same influences noted to act on typical SOC could act on the previously uncounted SOC (the non-soft-log C) giving a new potential emission of 5–5.4% of previously counted SOC. Biomass in primary-forests is concentrated in large trees, which, with secondary-forest growth (Kauppi et al., 2015) aside , are in global decline (Lindenmayer et al., 2012) and this could therefore be mirrored in the longer-term by decline in accompanying SOC.

For an approximation of the Extra-SOC that may be, or could be, in current forests, if one considers only the under-trunk Extra-SOC (60% of 6.9%) then the total area of global boreal, temperate and tropical forest of 32.7 Tm^2 as of 2000 (Hansen et al., 2010), combined with the SOC concentrations to -3 m under 'natural vegetation' of these forest types: 230(115) Mg ha^{-1} (Jobággy and Jackson, 2002), gives 31(1–72) Pg of Extra-SOC. To place this amount in perspective, in terms of global forest carbon, it is of the same order of magnitude as the global forest cover loss from 2000–2005 (3.1%) (Hansen et al., 2010). On an absolute basis, it is of the same order of magnitude as $1/20^{th}$ of the carbon in peat (Yu et al., 2010), and $1/10^{th}$ the carbon in live biomass of global forest cover (Pan et al., 2011). The existence of higher SOC in remnant primary forests suggests that there are higher emissions from past forest disturbances than have been inferred. Based on examination of remnant primary

forests, one could extrapolate to find the historical forest carbon stocks, taking into account that remnants may represent selection bias (Lindbladh et al., 2013). Such legacy Extra-SOC has either been emitted or is in the process of being emitted from the forests cleared from Neolithic times to ~1000AD (Kaplan et al., 2010), and it will be either contributing to global warming or have been re-sequestered in for example, the oceans or recovering forests— contributing to possible future sink saturation.

Chapter 7 Conventional intensive logging causes a long-term efflux of soil organic carbon from the mineral soil

This chapter is adapted from a published paper: Dean et al. (2016).

7.1 Abstract

There are few data, but diametrically opposed opinions, about the impacts of forest logging on soil organic carbon (SOC). Reviews and research articles conclude either that there is no effect, or show contradictory effects. Given that SOC is a substantial store of potential greenhouse gasses and forest logging and harvesting is routine, resolution is important. Forest logging SOC studies are reviewed and an overarching conceptual explanation for their findings is provided. The literature can be separated into short-term empirical studies, longer-term empirical and long-term modelling. All modelling that includes major aboveground and below-ground carbon biomass pools shows a long-term (i.e. ≥ 300 years) decrease in SOC when a primary or oldgrowth forest is logged and then subjected to harvesting cycles. The empirical longer-term studies indicate likewise. With successive harvests the net emission accumulates but is only statistically perceptible after centuries. Short-term SOC flux varies around zero. The long-term drop in SOC is driven by the biomass drop from the primary forest level but takes time to adjust to the new temporal-average biomass. Agreement is shown between secondary forest SOC stocks derived purely from biomass information and stocks derived from complex forest harvest modelling. Thus, conclusions that conventional harvests do not deplete SOC have been a function of their short time frames. Forest managers, climate change modellers and environmental policy makers need to assume a long-term net transfer of SOC to the atmosphere when primary forests are logged then undergo harvest cycles. However,

from a greenhouse accounting perspective, forest SOC is not the entire story. Forest wood-products that reach landfill, and some portion of which produce some soil-like material there rather than in the forest, could possibly help attenuate the forest SOC emission.

7.2 Introduction

Conversions of primary to secondary forests followed by long-term industrial usage are widespread (Rodgers, 1941; Knuchel, 1953; Foster, 1995; Hurtt et al., 2011). As the scientific results from studying logging processes are important in climate change modelling and are used in policy determination (IPCC, 2003; Sonne, 2006), the correct interpretation is important in climate change mitigation. There are recent claims of a consensus that soil organic carbon (SOC) does not change as a result of conventional logging (Achat et al., 2015; Hoover and Heath, 2015). Yet, a significant drop (Nave et al., 2010; Zummo and Friedland, 2011; Vario et al., 2014; Noormets et al., 2015; Petrenko and Friedland, 2015), a significant increase (Johnson and Curtis, 2001), and no consistent trend (Nave et al., 2010; Clarke et al., 2015) are concluded by other reviewers or from other data sets. There have been more short-term than long-term studies. One recent paper that concludes that conventional biomass harvests preserve the soil organic carbon (SOC) of forests relies on a global meta-analysis of 2028 values for SOC change dating up to 135 years after logging (Achat et al., 2015). The term 'conventional' here means not whole-tree-harvesting', i.e. at least the stump, roots and some debris from higher up the tree remains on site.

The focus of this chapter is on soil organic carbon (SOC) in the mineral soil (solum plus undifferentiated, weathered parent material), which often inadvertently includes for example, dissolved organic carbon (DOC). SOC includes soil-like material such as lignomor, which is decomposed, structure-less wood (Green et al., 1993). Away from the forest, SOC can include decomposed organic material in landfill technosols, some of which may be derived from forest wood (IUSS, 2014). However, except

where stated otherwise, the term SOC is used as defined in cited work, which can include both the mineral soil and the forest floor organic layer.

In the present work I argue that the controversy over the effects of logging is a function of the brevity of the timespans covered by the metadata, as the depletion of SOC after logging of primary forests is a matter of millennia, not decades. One determinant in the argument comes from the causative relationship between biomass and SOC observed globally across a range of ecosystems (e.g. Brown et al., 1993; Frank et al., 2012) and its implications for the long-term drop in biomass on conversion of primary forest to secondary forest (Harmon et al., 1990; Liski et al., 1998). That relationship is examined mathematically. Additionally, modelling from Chapter 4 was rerun so as to provide explanatory graphs on carbon dynamics, and to theoretically simplify contributions to change in SOC by removing the effects of intense silvicultural burns. In developing the proposition the results of previous studies on both short- and long-term changes in SOC with logging are summarised, before justifying developing equations between SOC and biomass, and between SOC and time since disturbance, and applying them to an example. Finally, earlier conclusions in the light of the analyses are critically assessed.

7.3 Short-term variation in SOC flux

The direction of SOC flux within the first 30 years after logging depends on a variety of contingent circumstances, including moisture, temperature, dead vegetation, soil type, tree species, degree of canopy closure, harvesting methods, lateral carbon transfer, degree of mechanical vertical mixing of soil, and degree of soil compaction (Johnson and Todd, 1998; Johnson and Curtis, 2001; Clarke et al., 2015). Following logging there can be a very short-term increase in SOC due to mixing of litter and soil during logging, or a very short-term decrease due to burning or exposure; both can be followed by a short-term decrease due to reduced biomass input, and then a medium-term recovery as the stand approaches maturity (Jurgensen et al., 1997; Chapter 4; Johnson et al., 2010).

Within one logging cycle, these complexities tend to produce a curve of variation of SOC with time approximating the displacement in damped simple harmonic motion (Peltoniemi et al., 2004; Diochon et al., 2009; Achat et al., 2015 their Fig. 4) (Figure 7-1).

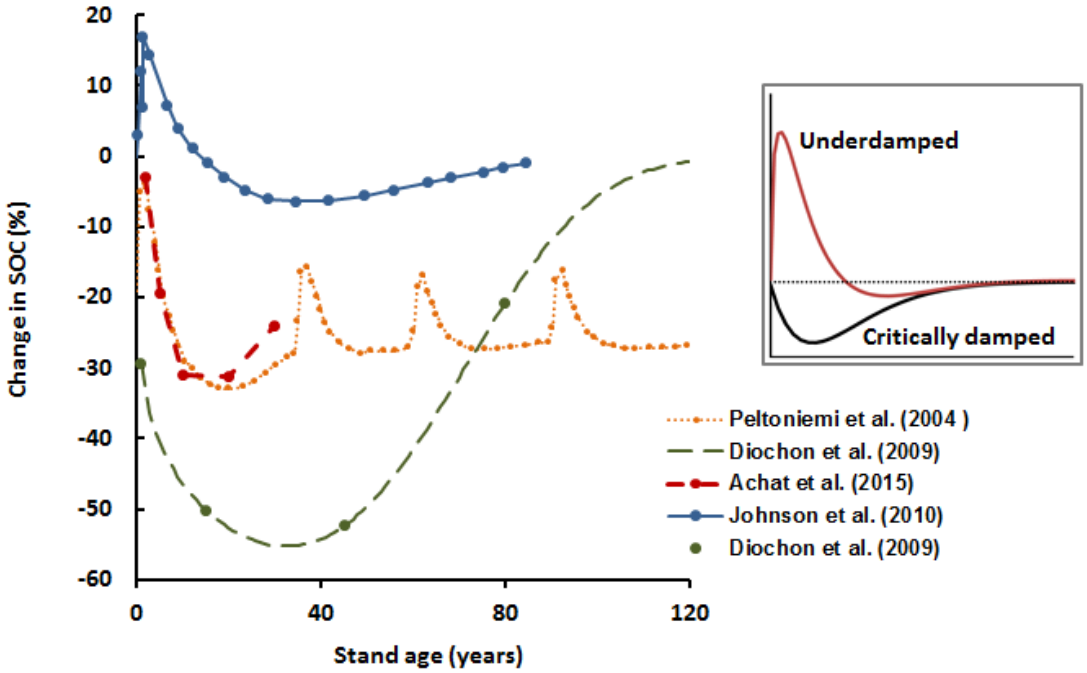


Figure 7-1. SOC variation with time within a single logging cycle, from empirical and modelling studies, showing similarity to simple harmonic motion (inset): a major decrease followed by an asymptote towards zero effect. Johnson et al. (2010) (Sugar Maple, CENTURY modelling, 40% biomass removed) was for mineral soil only so had a high initial peak due to incorporation of debris upon profile mixing. Peltoniemi et al. (2004) had thinning during recovery so shows multiple peaks. Achat et al. (2015) was for forest floor only so shows faster recovery.

The apparent direction of SOC flux can also be affected by depth of sampling, with the upper mineral soil responding more to change in litter and mixing during logging, than the lower horizons, which respond more to root turnover, including that of the coarse roots of just-logged trees and saplings undergoing self-thinning in the new stand. On average, apart from influences of priming, deeper SOC has a longer half-life than surface soils (Harkness et al., 1986; Fontaine et al., 2007; Frank et al., 2012; Braakhekke et al., 2013) and, therefore, will take longer to decrease from the effects of reduced biomass input. With the slow SOC pool vertically differentiated (Frank et

al., 2012), detection of change depends both on time and on depth, with change in lower profiles taking longer to become apparent.

7.4 Long-term decrease in SOC flux

Values for SOC change have been measured empirically, derived from mathematical simulation (modelling) or derived from a mixture of both approaches. Due to the short life-time of humans compared with the average half-life of SOC, long-term studies of change in SOC have been carried out by modelling. All modelling that includes major aboveground and below-ground carbon biomass pools and has more than ~40% aboveground biomass removal in each logging cycle, shows a long-term (i.e. ≥ 300 years) decrease in SOC when a primary or oldgrowth forest is logged then rotationally harvested (Table 7-1, Figure 7-2). Nevertheless, these models will be invalidated if long-term observations were to show different outcomes. Exceptions to the long-term SOC loss, i.e. an increase in SOC, were (1) Ranatunga et al. (2008) who did not include coarse roots in their modelling, and (2) (Johnson et al., 2010), who, using the CENTURY software with only 20% removal and a 120 year (i.e. long) cycle length had an SOC change of +0.7%, whereas the same parameters but with YASSO software found SOC change to be -3.2%.

Recently the nature of organic-mineral association and organic-particle-size for SOC has been revised (Kleber et al., 2015; Lehmann and Kleber, 2015). Half-lives of SOC haven not as yet been revised as a result of that work though the mechanisms revealed may suggest that more pools and consequently a greater range of half-lives, are necessary in models. That might change the shape of the SOC-stock-versus-time response within a logging cycle, but the existing shape has been confirmed empirically. That would imply that even though more half-lives may mimic reality better, the overall effect on SOC-versus-time may not change. Additionally, there is suggestion that half-lives may change differently with temperature than how some modelling of climate change impacts has suggested (Lehmann et al., 2007), but the

present work does not involve the change in temperature accompanying climate change.

Short-term SOC losses are always apparent after primary forests are logged (Table 7-2). For the slow pool of SOC in temperate forests, extreme half-lives of 1,592 and 245 years have been derived (Frank et al., 2012; Braakhekke et al., 2015). These rates of decline in the slow pool mean that it is highly unlikely that differences in SOC between successive individual rotations would be statistically perceptible from samples of the noisy forest edaphic environment. Most of the literature based on empirical studies attempts to make conclusions from such invalid comparisons. Many studies have only examined the early, variable oscillatory period (Figure 7-1), possibly inadvertently.

The modelling studies do not include error margins in the forecasts. Instead they rely on sensitivity analyses, observed biomass and decomposition dynamics, findings and common assumptions about SOC dynamics, and reproduction of observed total SOC dynamics (Peckham and Gower, 2011; Chapter 4; Wang et al., 2014). Comparing percentage change rather than absolute values cancels errors in absolute values of baseline total SOC stock and allows comparisons between different studies (Figure 7-2).

7.5 Dependence of SOC on biomass

For forested land in general there is a positive correlation between tree biomass and SOC. With conversion of primary forests to secondary forests on a harvesting cycle there is a drop in the temporal average biomass, which could therefore be expected to eventually translate into a drop in SOC. The end point of the decline in SOC after logging of primary forests and successive rotations is controlled by the relationship between biomass production and returns to the soil (Harmon et al., 1990; Liski et al., 1998). For Australian forests, using national-scale spatial data sets for SOC to 0.3 m

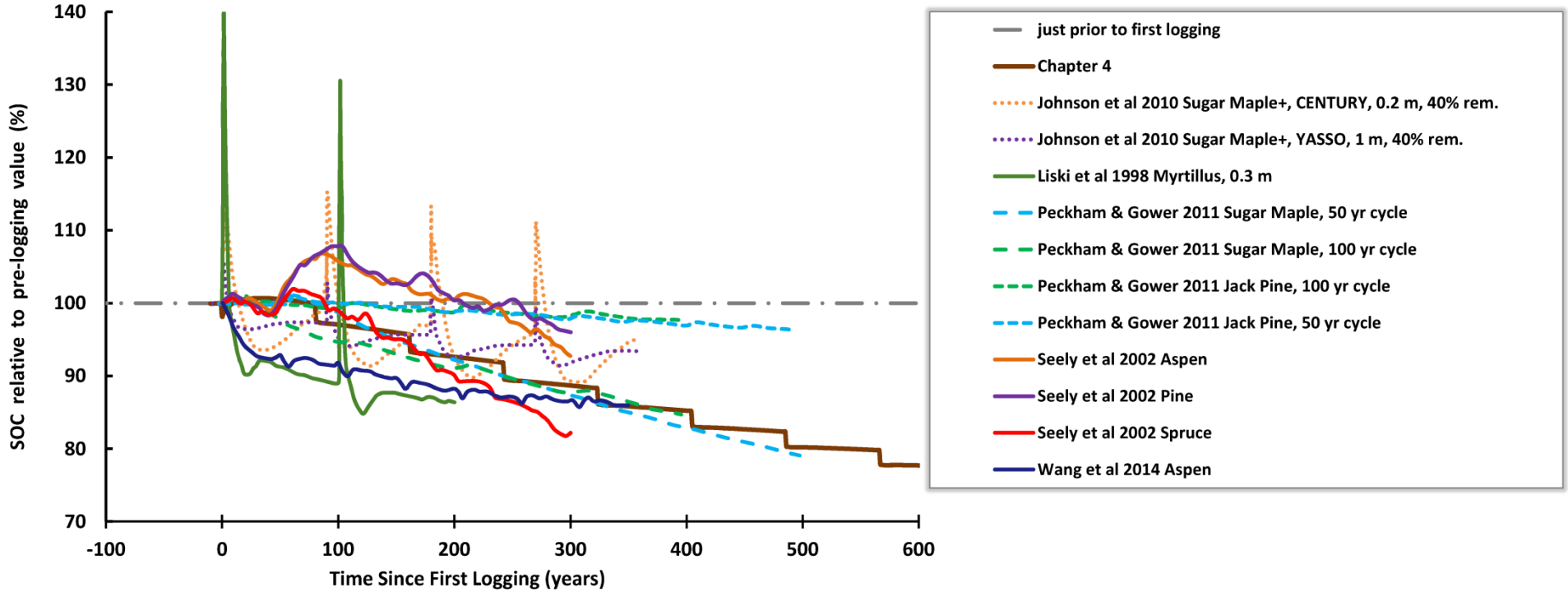


Figure 7-2. Long-term trends from modelling of SOC change with conversion of primary-forest to secondary forest with harvesting cycles. Data were extracted from figures in the literature and converted to percentage change compared with either the original primary forest, except for Peckham and Gower (2011) where they were compared with the non-harvest scenario. Sharper short-term peaks correspond to upper-level solum or extra included debris without burning. 'Chapter 4' shows succinct drops due to the intense burn of CBS. Shallower falloffs correspond to lower nutrient sites or higher biomass retention.

Table 7-1. Modelling values for net change in SOC (ΔSOC) after an extended period since initial logging (compared with Tables 7-2 and 7-3). The extended period allows the same effects as in Table 7-3 but more so, due to the longer period visible via modelling. [a]: Multiple model average. [b]: All SOC in the solum plus weathered parent material and that transported into the C horizon, and dissolved organic carbon (DOC) originating from the stand's biomass were included in modelling. Other models may have also included DOC, although unstated.

Reference	Cycles	Depth (m)	Time since 1st logging (years)	Forest type, logging type	ΔSC (%
Johnson et al. (2010)	4	0.2	360	hardwood, northern USA, no burn, 90% aboveground biomass removal	-1
Liski et al. (1998)	2	0.3	200	Finland, clearfell, no burn	-1
Liski et al. (1998)	20	0.3	2000	Finland, clearfell, no burn	-2
Johnson et al. (2010)	4	1	360	hardwood, northern USA, no burn, 90% aboveground biomass removal	-1
Peckham and Gower (2011, Figs 5 & 7)	10	whole mineral soil	500	northern USA, jack pine & maple average, clearfell, 50yr cycles	-1
Peckham and Gower (2011, Figs 5 & 7)	10	whole mineral soil	500	northern USA, jack pine & maple average, clearfell, 100yr cycles	-9
Seely et al. (2002, Fig. 6)	5	whole mineral soil	300	Boreal, aspen, spruce & pine average, 60 year cycles	-1
Seely et al. (2002, Fig. 6)	4	whole mineral soil	300	boreal, aspen, spruce & pine average, 75 year cycles	-9
Wang et al. (2014)	7	1.5 m, mineral soil	350	northern USA, aspen	-1
Chapter 4, Figure 4.4.a	3	whole profile & DOC[a]	240	wet-eucalypt, Australia, clearfell, burn & sow 2.5% loss	-1
Chapter 4, Figure 4.4.a	4	whole profile & DOC[a]	320	wet-eucalypt, Australia, clearfell, burn & sow 2.5% loss	-2
Chapter 4	19	whole profile & DOC[a]	1,500	wet-eucalypt, Australia, clearfell, burn & sow 2.5% loss	-4
present work	19	whole profile & DOC[a]	1,500	wet-eucalypt, Australia, clearfell, burn & sow 0% loss	-3

Table 7-2. Literature values of ΔSOC shortly after logging. Note that where change was only reported in the literature to a very limited depth, then SOC may have also changed deeper down.

Reference	Cycles	Depth (m)	Time since logging (years)	Forest type, logging type	ΔSOC (%)
Ellis and Graley (1983)	in 1st	0.02	0.005	wet-eucalypt, Tasmania, CBS, less-disturbed sites	-37.5
Attiwill and Adams (2008)	–	whole profile/ 0.03	-	wildfire, no logging, eucalypt, Australia, no soil disturbance	~ -2 / (-10 Mg ha^{-1})
Pennington et al. (2001)	in 1st	0.05	0.75	wet-eucalypt, Tasmania, CBS	-17
"	in 1st	0.05	"	"	-6
Tomkins et al. (1991)	in 1st	0.1	0.005	eucalypt, Australia, CBS	-20
Rab (1994)	in 1st	0.1	0.4	wet-eucalypt, Australia, no burn	-31[a]
Rab (1996)	in 1st	0.1	0.3	wet-eucalypt, Australia, CBS	-40[a]
Antos et al. (2003)		0.1	0.75	Douglas Fir, CBS	-26
Jiménez Esquilín et al. (2007)	in 1st	0.15	1.25	Ponderosa Pine, CBS	-50
"	"	"	"	under slash heaps	-16
"	"	"	"	edge of slash heaps, un-decomposed and partially burnt biomass	-20

Table 7-3. Empirically derived values for ΔSOC after an extended period since initial logging (compared with Table 7-2). The extended period allows: (a) decomposition of coarse plant matter disturbed during logging and hence contributing to SOC, (b) reduced long-term inputs to SOC from a lower temporal average biomass than in primary-forest, and (c) more differentiation from the pre-logging SOC level due to accumulation of effects from each contributing harvesting cycle.

Reference	Cycles	Depth (m)	Time since 1st logging (years)	Forest type, logging type	ΔSOC (%)
Rab (2004)	in 1st	0.1	10.4	wet-eucalypt, Australia, no burn	-48[a]
Hopmans et al. (2005)	in 1st	0.3	9	wet-eucalypt, Australia, CBS	8.5
Ferré et al. (2005)	3–4	0.45	40	Italy, plantation, no burn	-25
Tang et al. (2009, Fig. 4)	≥1st	0.6	73	hardwood, northern USA, no burn, clearfell	-21
Zummo and Friedland (2011)	in 1st	0.6	3	hardwood, northern USA, no burn, clearfell equivalent	-6
"	in 1st	0.6	55	"	-34
Vario et al. (2014)	≥2nd	0.6	<100	"	-24
Law et al. (2001)	in 1st	1	15	Ponderosa Pine, clearfell	-27

depth (Barson et al., 2002), potential aboveground biomass (AGB) (Richards and Brack, 2004), and GIS, I deduced the following relationship between SOC and AGB:

$$SOC= 188.7AGB/(298.8+AGB) \quad \text{Eq7-1}$$

where mass is in units of Mg ha^{-1} and $R^2= 0.38$, probability (P) < $5x10^{-8}$, N= 677.

Globally, a positive exponential correlation exists between SOC to 0.1 m depth and annual net aboveground biomass production (mass acquired per unit area per unit time) (Frank et al., 2012)

$$SOC=0.0715NAP^{0.332} \quad \text{Eq7-2}$$

where SOC is in Mg ha^{-1} and NAP is in Mg ha^{-1} $year^{-1}$, and $R^2= 0.78$, $P< 0.0005$, N= 13. A positive quadratic relationship exists between NAP and aboveground biomass (Keeling and Phillips, 2007) (except for high-productivity tropical forests when the relationship becomes negative), thus linking higher SOC with higher aboveground biomass for all except high productivity tropical forests.

Also globally, measured SOC is strongly positively correlated with modelled woody litter stocks (Potter et al., 1993) and woody litter is positively correlated with stand-level biomass in temperate forests (Harmon et al., 1986 data in their Table 10), thus linking SOC and aboveground biomass in temperate forests.

For tropical forests in SE Asia (Brown et al., 1993, data in Table 3), with accounting for land-use as of 1980, SOC to 1 m depth is related to below-ground biomass (BGB) by:

$$SOC=4.67BGB+ 61.7 \quad \text{Eq7-3}$$

where mass is in units of Mg ha^{-1}, and $R^2=0.39$, $P< 0.01$, N=14. The same study reveals a strong correlation between BGB and AGB, which is reflected globally as a root:shoot ratio (Mokany et al., 2006), thus linking higher SOC with higher aboveground biomass for SE Asian tropical forests generally.

For semi-arid woodland and forests in northern-eastern Australia, Harms et al. (2005) found a correlation of SOC to 0.3 m depth, with stand basal area (SBA), which in turn for those ecosystems, is linearly proportional to aboveground stand biomass (Burrows et al., 2000):

$$SOC = 2.58SBA + 5.39 \qquad \text{Eq7-4}$$

where mass is in units of Mg ha^{-1} and SBA is in m^{-2} ha^{-1}, and $R^2 = 0.57$, $P < 0.01$, N= 28. Basal area for that region is strongly positively correlated with aboveground biomass (Henry et al., 2002), thus again linking higher SOC with higher aboveground biomass.

These measured relationships between biomass and SOC are not primarily related to land use change, but rather to climate, fire regimes, and soils. They unavoidably incorporate some degree of influence of land-use. Note that these relationships generally exclude extreme ecosystems such as tundra, where the positive correlation between biomass and SOC may break down, as SOC storage there is perhaps more dependent on lack of heat than on plant productivity (causing disequilibrium between C inputs to soil and C outputs, resulting in long-term net accumulation). In the absence of long-term empirical data for a purely land-use relationship for forests, these multi-influence data shall be used to model change in SOC with logging.

7.6 Time dependence of SOC stock— effective SOC half-life

In order to calculate the duration after logging from whence a measurable change in SOC has occurred, the overall decomposition (mineralisation) rate of SOC must be considered. For this process one uses the analogy of radioactive decay (Jenny et al., 1949; Chapter 8; Olson, 1959), which includes the concept of decay chains comprised of ‘daughter’ pools derived from ‘parent’ pools. From modelling and experimentation with radiocarbon data, Braakhekke et al. (2015) found for a temperate deciduous forest in Germany, that the half-life for the slow (i.e. slowest or ‘passive’) SOC pool was 1,592 years, and that the slow pool occupied 50% of the

total stock from 0-to-0.3 m, but 91% from 0.3-to-0.6 m (Braakhekke et al., 2015 Fig. 3) (and it was therefore a major determinant). The parent pool in the decay chain had a half-life of 1.8 years and the fast daughter pool had a half-life of 29 years (Braakhekke et al., 2015 Fig. 2). Basic simulation of SOC flux using these parameters of a three pool system show that the system's SOC combined half-life for SOC efflux was close to the slowest one; i.e. it is a limiting half-life. In general, the time required for SOC of the entire solum plus weathered parent material to respond to change in long-term biomass, for example due to conversion of primary forest to secondary forest on a harvesting cycle, matches such long half-lives (Liski et al., 1998; Chapter 4). Although half-lives may change when new nutrients are added due to the priming effect, for example by sudden coarse root death, such effects have not been quantified for the logging scenario. Therefore half-lives were not adjusted for the priming effect in the present work.

Parton et al. (1988) deduced a turnover time of 800–1200 years (i.e. a half-life of 554 to 832 years) for the slow SOC pool of the USA Great Plains grasslands. Their average half-life of 693 years was used globally (Mills et al., 2014) and for *E. regnans* forests in Australia (Grierson et al., 1991; Chapter 4; Dean et al., 2003). For a temperate forest in England, Harkness et al. (1986) found the half-life of the slow pool to be 350 years. From global radiocarbon studies Frank et al. (2012) calculated a half-life of 245 years for the slow pool of mineral soil to 0.1 m depth in a USA northern hardwood forest. The values of Braakhekke et al. (2015) and Frank et al. (2012) were used in the basic simulations in the present work.

7.7 Logging and SOC in Eucalyptus regnans forest

The *Eucalyptus regnans* tall hardwood evergreen forests is used here as an exemplar of SOC change after logging of primary forest to reveal explanatory detail of SOC change. For conversion of primary *E. regnans*-dominated forest to secondary forest on an 80 year harvesting cycle, the temporal average AGB drops by 63% (Chapter 4). For a typical forest in the Styx Valley of Tasmania that means a drop in

temporally averaged AGB from 740 to 275 Mg ha^{-1}. That decrease in biomass and the link between biomass and SOC from Eq7-1 were used to model ΔSOC. This was calculated for a slow SOC pool half-life of 1,592 years (Braakhekke et al., 2013) and one of 245 years (Frank et al., 2012) to determine ΔSOC with each harvesting cycle and when it would be detectable. Secondly, the previous modelling of logging and harvesting (Chapter 4) was re-run but with zero ΔSOC during the logging operations, to test the specific influence of the 2.5% SOC loss from operational-specific disturbances and regeneration burning during clearfell, burn and sow (CBS).

A long-term (~1,500 years) drop in SOC (Figure 7-3, 7-4) of 39% was forecast by the first simulation. This percentage does not compare well with the earlier modelled drop of 48% (Chapter 4), but the latter included the additional loss of 2.5% accompanying the intense regeneration burn with each CBS event. In the modelling without SOC loss due to silvicultural burns, SOC still declined in the long-term, but by only 35%, compared with 48% when the CBS effect was included. This is in good agreement with the basic simulation (39%), confirming that reduction in biomass is the cause of the long-term decrease in SOC. The difference between the 35% and the 48% means that just over a third of the long-term loss of SOC is due to losses during CBS, and approximately two thirds due to a temporally averaged lower biomass in the secondary forest than in the primary forest.

If the empirical error margin on measured SOC stocks is 5% (which is optimistically small), and twice the error margin is required to confirm ΔSOC, then no ΔSOC will ever be visible between successive harvesting cycles, as even the largest net change is less than 10% of the SOC stock (Figures 7-3, 7-4).

On conversion of primary to secondary forests, the change per cycle is initially large then decreases exponentially. After ~1,000 years the change for a single cycle may be undetectable (Figure 7-4). The difference compared with the original primary forest continues to increase, and it is that change which is likely to be most readily detected. For example, for the scenario shown in Figure 7-4, at 500 years (1 primary

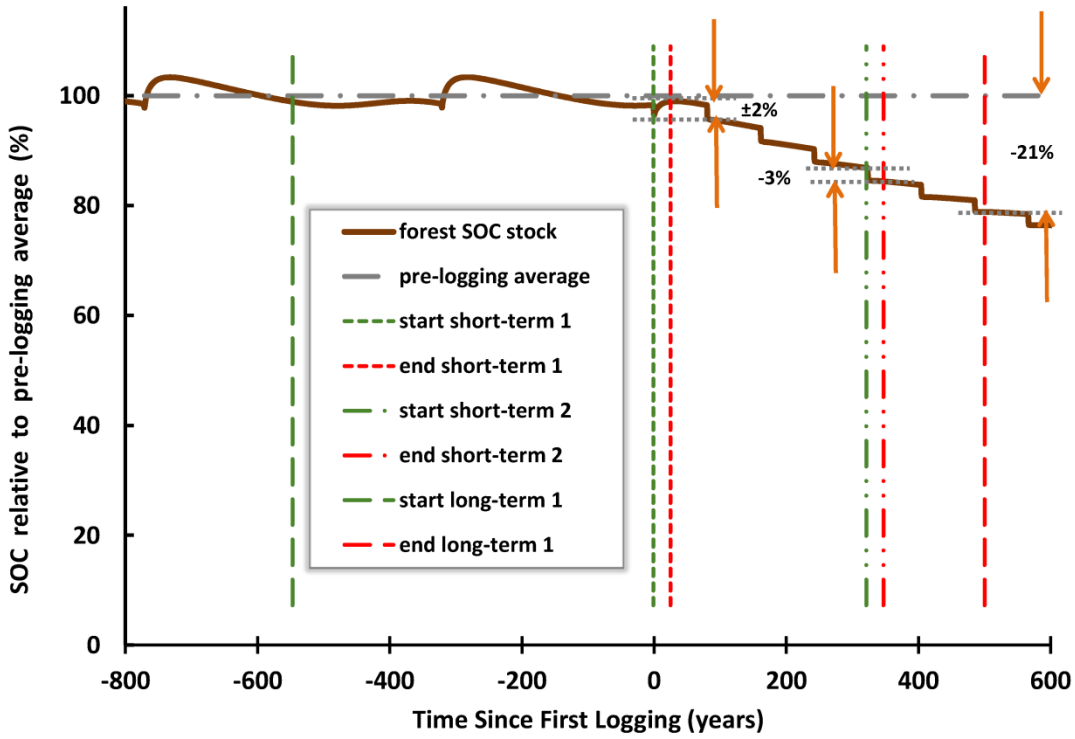


Figure 7-3. Total SOC pool following conversion of primary forest to secondary forest on 80 year harvesting cycles with clearfell, burn and sow silviculture. Individual changes with the initial logging or harvest may not be distinguishable but the accumulated change over several cycles should be detectable empirically.

forest logging and 6 secondary forest harvesting cycles) the cumulative ΔSOC is -6% and -26% for SOC with half-lives of 1,592 and 245 years respectively. The modelled SOC with the longer half-life, had no net visible change for 880 years but with the shorter half-life net SOC change was visible after 240 years. These durations are in good agreement with the 200 years found necessary to detect change in Scotland (Miller et al., 2001).

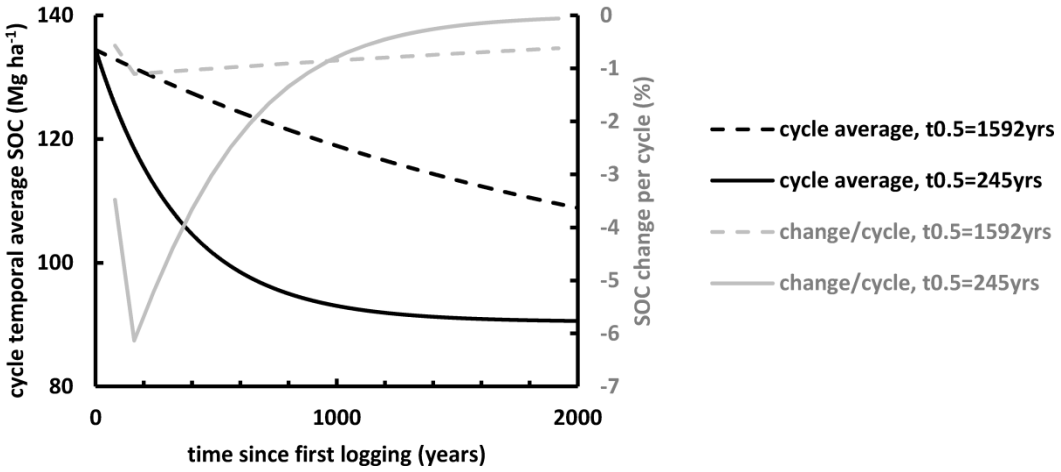


Figure 7-4. Slow SOC pool following conversion of primary forest to secondary forest on 80 year harvesting cycles, with two different slow SOC pool half-lives. Drops in SOC between successive harvests decrease exponentially. If comparing the percentage difference with primary forest then the cumulative percentage change is the meaningful amount. Fluctuations within individual harvest periods are not shown.

7.8 Revisiting the literature

It is particularly difficult to get an accurate measure of SOC in primary forests due to the high spatial variability. High sample variance increases the number of samples required in order to detect change within a reasonable time period (Poussart et al., 2004). The variation is likely to be greater than already found, as sampling underneath, and in close vicinity to tree trunks is not done, for obvious reasons. Yet, it is in these locations that the highest levels of soil carbon inputs could be expected (Lutz, 1960; Liski, 1995; Throop and Archer, 2008; Rossetti et al., 2015). Additionally, the detection of change must be made against the background noise of short-term fluctuations due to soil disturbances during logging (e.g. Yanai et al., 2003a; Rab, 2004). The SOC profile in the mineral soil can depend on the tree species present (Ahmed et al., 2016). Therefore if the species composition is different in the secondary forest, comparison experiments are more likely to need more thorough sampling. The long-term modelled ΔSOC for some typical silviculture types, is ~10%, which may be difficult to detect (Johnson et al., 2010).

Conversely, shorter harvest cycles and less retention of debris on-site will hasten the long-term SOC loss (as temporally averaged biomass is lower) and make change between harvests greater and more discernible (Johnson et al., 2010).

Modelling shows a peak in SOC within approximately the first few decades after logging, due to input from logging debris and freshly killed roots (Liski et al., 1998; Seely et al., 2002; Peltoniemi et al., 2004; Johnson et al., 2010; Dean et al., 2012d) (Figures 7.1–7.3). In models the smaller debris may remain in the debris pool rather than in the SOC pool (Chapter 4), and thereby provide contrast between model forecasts and empirical results. Dissociation of mineral carbonates with intense heat where intense regeneration burns are present, would also provide contrasts between modelling and experimentation, though in the other direction.

The short time span used in much experimentation could be one reason for the difference between the results of Hopmans et al. (2005) and Rab (2004), for example, with the deeper levels in Hopmans et al. (2005) showing some SOC increase due to decomposition of buried, unburnt debris.

All empirical results showed a long-term loss of SOC where there had been two or more cycles (Table 7-2). The long-term decrease was echoed in modelling studies where all major above and below-ground carbon biomass pools were modelled and 40% or more of aboveground biomass removed in each logging cycle (Table 7-3, Figure 7-2). Ranatunga et al. (2008) is an exception to this finding, but they did not include coarse roots.

The finding of a decrease in the upper mineral soil (0–0.1 m) and an almost compensatory increase in the lower mineral soil in Achat et al. (2015) can be explained in terms of the interaction between coarse root location and short-term experimentation: coarse roots are likely to be concentrated below 0.1 m and take longer to decompose than the surface litter and debris mixed during logging in the upper mineral soil, and most of experiments examined in the review had only just covered the preliminary stages of the below-ground tree decomposition and regrowth. Their net -6% ΔSOC is mainly a short-term result, i.e. short considering

the limiting half-life of soil carbon and the multitude of influences impacting on recently logged forest soils.

The finding given in Noormets et al. (2015) from the SRDB database (Bond-Lamberty and Thomson, 2010) (version 20120510a) that SOC stocks in unmanaged and 'natural' forest are larger than those in managed forests, was reassessed using the latest version of the database (version 20150826a). The finding could not be replicated as the standard deviations on categorised data were too large to allow categories to be differentiated. The SOC stocks reported in the database are for a range of depths, with some records not indicating depth of measurement. Also some SOC records are duplicates, differing in, for example, soil respiration or fine root mass measurements. Sorting on the basis of management type ('managed', 'unmanaged' and 'natural'), growth stage ('aggrading' or 'mature') and SOC depth measurement, and removing duplicates, left 363 records, and 199 with depth measurements. This small number of observations resulted in no significant difference in SOC stocks for different management categories, not even for specific depths: 117(112) Mg ha^{-1}, 114(100) Mg ha^{-1}, and 133(174) Mg ha^{-1} , for 'unmanaged', 'managed' and 'natural' forests respectively across all depths.

One possible trend, although not statistically significant, was that depth measurement was greater for 'unmanaged' forest compared with 'managed' and 'natural' (0.80(0.38) m, 0.44 (0.25) m and 0.45(0.27) m respectively). Cumulative SOC increases with depth, so if this depth trend had been present in the 2012 version of the SRDB examined by Noormets et al. (2015) then that could have yielded their observation (rather than the influence of management). Another possible trend, though also not statistically significant, was for 'aggrading' forest to have higher SOC stocks than 'mature' forests but to be measured to a greater depth. This is possibly because the roots of mature trees are likely to obstruct SOC measurement. Other confounding factors were that: few sites were environmental pairs across management types, some managed forests were categorised as natural, and some records had been transcribed from the original report into the SRDB with incorrect unit conversions e.g. for Deng et al. (2012). Many more data points with known

depth of measurement would be required for discrimination of management effects from a database that does not have site pairing.

With regards to the multiple-cycle, long-term effects explained above I have found another reason apart from the priming effect suggested by Noormets et al. (2015) for the decline in SOC with logging. If both operate then the effects will be additive. Losses at the time of logging (e.g. due to the intense fire of CBS silviculture) and any possible long-term nutrient or physical nutrient-holding ability associated with logging e.g. Gough et al. (2007a) will constitute further attrition of SOC.

Many forests in Europe and North America have already been secondary forests for more than two centuries. Consequently, as the drop in SOC between successive logging cycles decreases exponentially (Figure 7-4) then change is unlikely to be detectable between successive harvests in these areas. Some nearby primary forest with which to compare the effect of logging on SOC would be needed to measure the impact.

In their critique of Zummo and Friedland (2011), Hoover and Heath (2015) note that the mature forest has 9% less SOC to 0.6 m solum depth than the post-logging, low category logging-disturbance forest. They use this to support their conclusion that there is no significant drop in SOC due to logging. However these data do not support such a conclusion because:

1) The post-logging area is a mixture of low-, medium- and high-disturbance (not just low). The average SOC change, assuming equal areas of each disturbance type is -5%. But the majority of the area is likely to be of medium-disturbance, which had a SOC change of -6.6%. The high-disturbance portions had -21%.
2) This is the 3 year post-logging data (i.e. still in the fluctuating flux-direction stage), not the 55 years post-logging, which had higher losses to 0.6 m depth: 3 year old low-disturbance 93.3±4.9Mg C ha^{-1} , 55 year old 56.3±4.2Mg C ha^{-1}, mature 85.1±4.4Mg C ha^{-1}.

3) The mature forest at the 0.45–0.6 m depth had 46% more SOC than the average logged area, and this horizon could have been deeper in several spots— increasing calculated SOC loss on logging.

In the meta-analysis of Nave et al. (2010) there was a finding of an overall absence of change in SOC in the mineral soil for logging temperate forest but a 9% drop for hardwood forests. The data in Nave et al. (2010, Fig. 3) show a SOC loss up until ~25 years for average Alfisol, Inceptisol and Ultisol soils, and up to ~70 years for Spodosol soils. The work of Nave et al. (2010) examined only short-term change (maximum of 80 years) and not multiple-cycle effects. Therefore their conclusions do not refer to long-term effects of logging on SOC. Hoover and Heath (2015) refer to the complexity and different findings regarding SOC change with logging, as reviewed in Buchholz et al. (2014), and declare that that diversity is sufficient reason to concur with a so-called 'consensus' on no net effect. However, it is possible to partition existing metadata sets in ways that allow different propositions to be tested. For example, the effects of logging secondary forest could be contrasted with the effects of logging primary forest.

Although no change in SOC is as yet detectable up to ~50 years since first disturbance for many Brazilian tropical forests where there is a 40% loss in biomass (Berenguer et al., 2014), the present work suggests there is likely to be around a 30% loss in SOC in the long-term. Globally, with warming due to climate change, SOC half-lives for the upper 0.2 m of the solum are decreasing (Chen et al., 2012). This implies a need to conserve that SOC for as long as possible. One way to enable climate change mitigation is therefore to discontinue the logging of secondary forests, especially those with less than a few cycles since they were primary forests.

Norris et al. (2011) found that mature primary-forests with higher biomass have cooler soil temperatures than managed secondary forests of the same species. Using the negative exponential relationship between SOC-half-life and mean annual temperature of Frank et al. (2012):

$$SOC\_turnover\_time = 703 * \exp(-0.103 * MAT) \qquad \text{Eq7-5}$$

where *SOC_turn_over_time* is in years and *MAT* is in degrees Celsius, R^2= 0.95, P<0.0001, would indicate longer half-lives for oldgrowth forests. This arises due to the exponential relationship causing inequality of impacts of the temperature differences between forest types and across seasons within a year. This potentially adds to the other thermodynamic amelioration reasons given in Norris et al. (2011) for conservation of mature primary forests, for climate change mitigation.

From a greenhouse accounting perspective, forest SOC is not the whole story. The decomposition of forest wood-products that reach landfill (e.g. used lumber and paper) will contribute in part to SOC in the landfill rather than in the forest, which could possibly help attenuate the net forest SOC emission from logging. Most comprehensive models of forest carbon dynamics include both SOC and the fate of wood-products. But, when calculations include the landfill SOC, a full life-cycle-analysis (LCA) of the forest timber is required. This LCA would require more approximations, such as diesel usage in transport, paper recycling-mill emissions, public use of wood-products and mill residues as fuel, possible priming effect in landfill, the more-polluting methane, and further mechanical disturbance. This is discussed further in section 4.5.3. Silvicultural interventions such as fertiliser addition to the forest would possibly alter the forest carbon balance but then one would also have to include the LCA of the fertiliser addition.

In summary, the temporal attribute provides a simple explanation for earlier perceived differences in results from experiments on the effect of logging on SOC, and a way to portray those individual findings such that they fit a common theme and a common result. Primary forest must be distinguished from secondary forest and as the latter undergo more harvesting cycles the SOC difference between cycles will become more indistinguishable, though the SOC in repeatedly harvested forest will become more distinguishable from the SOC of the primary forest. Forest managers, climate change modellers and environmental policy makers need to assume a long-term net transfer of SOC to the atmosphere when primary forests are logged then undergo harvest cycles. This includes calculations for primary forests that were

logged and converted from decades to millennia ago, and whose emissions are now contributing to current climate change.

Chapter 8 Accounting for space and time in soil carbon dynamics in wooded rangelands

This chapter is adapted from a published paper: Dean et al. (2012b)

8.1 Abstract

Employing rangelands for climate change mitigation is hindered by conflicting reports on the direction and magnitude of change in soil organic carbon (ΔSOC) following changes in woody cover. Publications on woody thickening and deforestation, which had led to uncertainty in ΔSOC, were re-evaluated, and the dimensional-dependence of their data was determined. To model the fundamentals of SOC flux, linked SOC pools were simulated with first-order kinetics. Influences from forest development timelines and location of mature trees, with a potential for deep-set roots, were considered. It is shown that controversy or uncertainty has arisen when ΔSOC data were not measured along sufficient lengths of the three Cartesian axes and the time axis, i.e. in 4D. Thickening and deforestation experiments have particularly neglected factors affecting the time and depth axes, and sometimes neglected all four axes. Measurements of thickening must use time-spans beyond the calculable breakeven date— when thickening just recovers the SOC lost through land degradation: then all ecosystems are likely to incur net sequestration. The similarity between half-life of carbon pools, and the half-time required for sequestration, mandates that millennial time-spans must be considered in design of SOC experiments. Spatial and temporal averaging of ΔSOC data that accounted for environmentally dependent decomposition rates, revealed that deforestation to pasture incurred a higher and longer-term net emission than earlier reported. Published reports on thickening or deforestation appear no longer contradictory when one considers that they only presented views from lengths of the

4D axes that were too limited. Adoption of this understanding into carbon accounting will allow more precise estimates of carbon fluxes for emission trading schemes and national reports.

8.2 Introduction

For many rangelands woody vegetation plays a major ecological role, through vegetation types such as: (a) shrubland, woodland, open-forest, riparian forests in grasslands, or savannah; (b) improved pastures after deforestation; or (c) extant grassland experiencing woody thickening; (e.g. Carnahan, 1977; Lund, 2007; Ellis, 2011). The term 'wooded-' rather than 'forested-' is used here with respect to rangelands, to avoid confusion between different countries' definitions of forests, and to include the woody-thickened state. The term 'wooded rangeland' has been used for rangelands worldwide that have a perceptible cover of shrubs, open woodland, riparian forests, or a savannah-like tree distribution (e.g. Meuret et al., 1993; Greene et al., 1994; Eldridge and Rath, 2002; McLeod, 2002; Papachristou et al., 2005; Milán et al., 2006; Wambui et al., 2013).

Soil organic carbon (SOC) is a substantial terrestrial carbon (C) pool, often positively correlated with plant biomass (Su and Zhao, 2003; Harms et al., 2005; Hughes et al., 2006; Wheeler et al., 2007). The correlation means that SOC in many rangelands can be influenced by: overgrazing and land rehabilitation; deforestation and regrowth; woody thickening; fire; and climate change (Batjes and Sombroek, 1997; Reeder, 2002; Smith and Johnson, 2004; Luo et al., 2007). Consequently, rangeland SOC can act as a C sink in support of greenhouse gas (GHG) mitigation projects (McKeon et al., 1992; Glenn et al., 1993; Walker and Steffen, 1993). However, the difficulty in measuring and forecasting rangeland SOC, and conflict between reports on ΔSOC, is related to timescales and measurement practices (section 1.1.2).

Conceptualisation in four dimensions (4D, the three Cartesian axes plus time) has provided scientific insight and management options, in ecology and C dynamics (e.g.

Ward, 1989; Dean et al., 2004; Dean and Roxburgh, 2006; Turnbull et al., 2008; Colyan and Ginzburg, 2010). Woody thickening in rangelands, has significant 4D diversity in SOC concentrations (Boutton et al., 1998). The hypothesis in this chapter is that conceptualising rangeland C dynamics in 4D can provide a coherent solution to dissonance over ΔSOC for significant rangeland processes. Two examples are selected for re-evaluation using this hypothesis, from reports where conclusions had important consequences for climate change mitigation.

The sampling span (extent, length) used in rangeland experiments is crucial. For example, a correlation between biomass and SOC has sometimes appeared ambivalent but was confirmed as positive when experiments were longer-term and when SOC was sampled at depths that incorporated more of the root extent under woody vegetation than otherwise (Groenenberg et al., 1998; García-Olivia and Masera, 2004; Liao et al., 2006a; Liao et al., 2006b). Longer-duration experiments on change in biomass or ΔSOC are generally more definitive than shorter ones (Lugo and Scatena, 1996; Diochon et al., 2009). Conversely, short-duration (decadal) experiments may yield low-magnitude and conflicting results for ΔSOC (Lugo et al., 1986; Guo and Gifford, 2002; Liu et al., 2004; Marin-Spiotta et al., 2009). Although SOC is generally correlated with vegetation cover there can be short-term fluctuations in SOC when the vegetation cover changes, which can mask the influence of interest. Examples include the priming effect (Fontaine et al., 2007), and early oscillations in SOC following deforestation, followed by long-term emission and an asymptote (García-Olivia et al., 1994; García-Olivia and Masera, 2004). SOC can take approximately two millennia to stabilize, such as for rangeland thickening (Hibbard et al., 2003) and for the decrease in SOC when forests were replaced by prairie due to fire during the Holocene (Baker et al., 1996).

Accounting for dimensions is also important during data processing. For example, when averaging samples, their equivalence with respect to the effect being studied is assumed, but that effect may change nonlinearly in a particular dimension. Therefore Jensen's inequality (Jensen, 1906; Welsh et al., 1988; Chesson, 1998; Ruel and Ayres, 1999) becomes relevant. It implies for a non-linear function that the average

of the outputs is not necessarily equal to the output of the average input. For example, site location, (environment), is a function of x-y position but SOC decomposition rates may vary between environments. SOC emission following deforestation is higher from higher SOC sites (Holmes et al., 2006) but proportionally higher from poorer soils (i.e. lower SOC sites) (Harms et al., 2005). Averaging data over such complexity blurs effects and thereby limits accuracy of findings. Here the influences over 4D of deep-set SOC, and deep-rooted, long-lived plants and their decomposition products are considered. The focus here is on the time and depth (z) axes as these have been the most neglected, but data from all four axes are integrated into the re-analyses.

8.2.1 Background to self-thinning in woody thickening

When discussing the timelines of woody thickening and the accompanying ΔSOC it is necessary to consider underlying ecosystem dynamics. After the onset of thickening, in situations where the environment has supported a high, initial stand density of seedlings and saplings, then stand density gradually decreases, due to competition for resources. The juvenile trees (or shrubs) (Figure 8-1) that are outcompeted die and survivors become larger— the biomass of both the survivors and the entire stand increases. Stand density usually decreases logarithmically. This process of stand development, known as 'self-thinning' forms a foundation of the mathematics and ecology common to forest science. Self-thinning is not often considered in carbon accounting or ecological studies in the rangelands but it has been observed in woody thickening in the rangelands (e.g. Felker et al., 1990; Boutton et al., 1998; Fensham and Fairfax, 2005). Additionally, size-age distributions, which fit the dynamics of self-thinning (although not identified as such) appear in several reports on rangeland shrubs and trees (e.g. Adams, 1967; Crisp and Lange, 1976; Eldridge, 1988; Watson and Holm, 1994; Zerihun et al., 2006; Krull et al., 2007). Drought, overgrazing and poor site quality can all reduce net growth and thereby stymie self-thinning (Horne and Robinson, 1987; Florence,

1996; Svejcar et al., 2008). In these cases the self-thinning process may stall or appear to be absent, and a strong perturbation may be necessary to initiate faster change in stand density (e.g. destocking, flood or mechanical damage (although some such perturbations may also precipitate a second thickening germination event)). Self-thinning may also be undetectable at some time scales for slow growing species (e.g. drought tolerant *Callitris sp*., Ross et al., 2008).

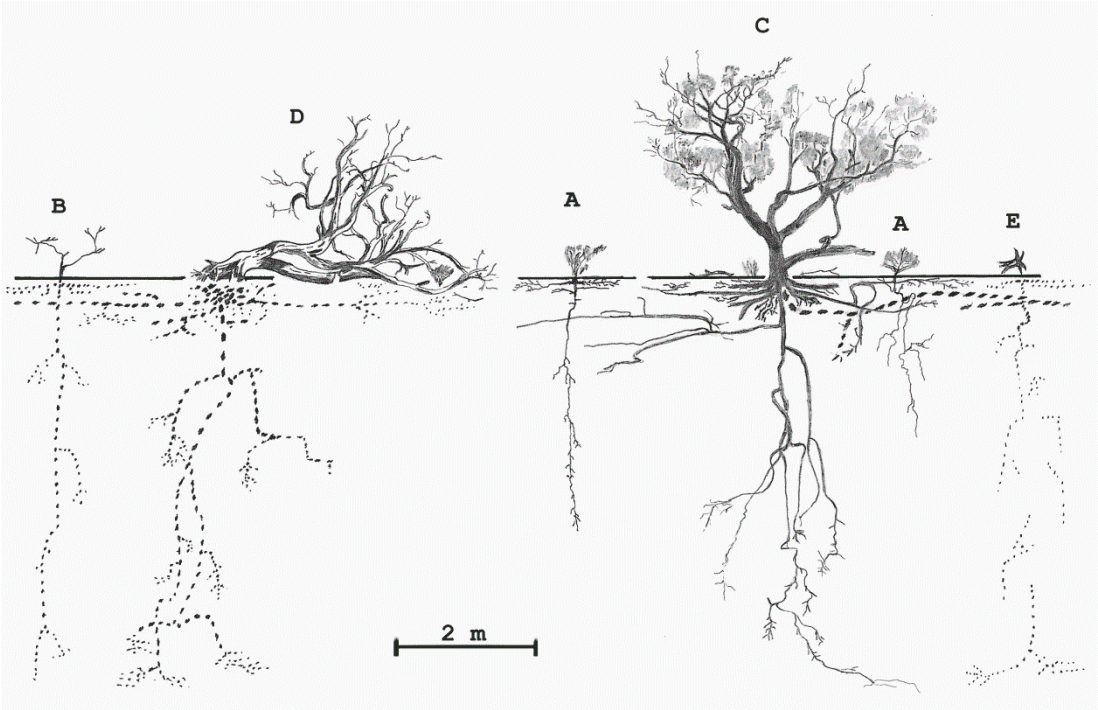


Figure 8-1. Four-dimensional distribution of biomass, its decomposition residues and SOC from different growth stages of a woody thickening species in semiarid rangeland. The SOC portrayed is that arising from root residues, prior to any eluviation or soil turbation. Dotted lines indicate decomposing roots and resultant sites of concentrated SOC (Chabbi et al., 2009); fainter dots indicate more advanced decomposition. Tree growth stages: A— juvenile, B— juvenile dead, C— mature/senescent; D and E— dead and fallen; E— stump (dead). For clarity, the larger number of lateral, near-surface roots, have been omitted, as have the larger number of stages A and B for young or overgrazed stands. With self-thinning, the older and larger individuals will be further apart in the x-y plane, creating a 4D trend in SOC concentration. Biomass decomposition is necessary over several generations of shrubs (or trees) to form the maximum SOC stock (*cf* Figures 8.2a-b and 8.4).

Self-thinning is more obvious where germination events are infrequent. This is because repeated germination events prior to stand maturity of the oldest age cohort will create (a) multiple self-thinning stands that overlap in both time and space, and (b) competition between different age cohorts. With temporal overlap, a mixture of mature, dead and early-stage-thickening may prevail (e.g. Grice et al., 1994; Fensham and Fairfax, 2005).

Three major effects of self-thinning that must be considered when measuring ΔSOC are: (a) SOC will be more concentrated where the roots of the thinned individuals once were (e.g. Chabbi et al., 2009); (b) higher concentrations of SOC will accrue after the larger individuals have senesced; and (b) it may take several cycles of germination and senescence for the maximum, stand-wide distribution of SOC to accumulate.

8.2.2 Background to the 4D distribution of SOC in wooded rangelands

The likely distribution of C in 4D, when the SOC is supplied by woody vegetation, forms a foundation for the interpretation of ΔSOC. To portray key features the biomass distribution of a semiarid thickening species (Figure 8-1) was idealised from the literature (Heitschmidt et al., 1988; Gile et al., 1995; Gile et al., 1997). The time-dependent changes illustrated are inherent in the natural growth and decomposition of woody vegetation, of which self-thinning is a key component (discussed below). From a SOC isotopic study of mesquite thickening in Texas (Boutton et al., 1998) self-thinning was found to be ongoing in the 1990s following land degradation in the previous century. Figure 8-1 illustrates that the concentration of SOC derived from aboveground carbon and from roots (Chabbi et al., 2009), varies over the x-y plane and with depth (i.e. in 3D). The depth concentration of SOC is likely to vary with tree maturity, and due to self-thinning the distance between individual trees increases with age, and thus so will the 3D variation of SOC, but with a time lag while the biomass decomposes. The distribution of biomass, and therefore of SOC, is

perturbed by land-use such as overgrazing, deforestation and reforestation. SOC flux is therefore a function of five variables:

$$\text{SOC flux} = F(m, x, y, z, t) \qquad \text{Eq8-1}$$

with units of $Kg.m^{-3}.s^{-1}$. I.e. the SOC flux is the change in 3D SOC concentration multiplied by the rate of change. For the pedosphere however C flux is usually reported in change of mass per unit area per unit time (e.g. Mg ha^{-1} $year^{-1}$) which, although correct for an area-flux, has possibly helped engender complacency in measurement depth. The IPCC recommends that measurements be taken down to where significant change is expected to occur (IPCC, 2000). This condition of 'significant' is ambiguous, with deviations from the original intent facilitated by (a) the 0.3 m span common in analysis of arable lands, (b) faster change in the upper horizons (e.g. Dalal et al. (2005)) perhaps being interpreted as the only change of significance, and (c) difficulty in sampling deeper or under lateral roots. Frequently these shortcomings are unaccounted for in data processing and in the conclusions drawn.

The relevance of sampling extent combined with the relevance of measuring in 4D, necessitates an appropriate sampling extent along each of the four axes. From a literature survey it appeared that to-date, ΔSOC in rangelands has only been measured in limited combinations of each of the four axes (within any particular experiment), e.g. in the x-y plane (the surface soil) at different scales (Reeder, 2002; Holmes et al., 2006; Wheeler et al., 2007; Throop and Archer, 2008), along the z-axis (Jackson et al., 2002; Fontaine et al., 2007), in x-y-z (Don et al., 2007; Chabbi et al., 2009), along the t-axis (i.e. temporally) (Hibbard et al., 2003; García-Olivia and Masera, 2004; Poussart et al., 2004; Chiti et al., 2009; Diochon et al., 2009; Schulp and Verburg, 2009), and in x-y-t (Wilson and Thompson, 2005). Details of appropriate sampling of SOC in the x-y plane have been previously described (e.g. Conant and Paustian, 2002; Poussart et al., 2004; Holmes et al., 2006; Allen et al., 2010). Temporal measurement of rangeland SOC has been discussed but only for seasonal affects (Allen et al., 2010), i.e. a short extent on t.

The SOC distribution along the z- and t-axes warrants comment. Below-ground biomass is expected to provide the major source of long-term SOC and dominate SOC dynamics in semiarid areas through root turnover and tree mortality (Wang et al., 1996; Lorenz et al., 2007; Filley et al., 2008). Carbon from deeper roots, can contribute more to SOC than does aboveground biomass (Rasse et al., 2005). Woody plants in semiarid areas may have a dimorphic root structure and be phreatophytic, combining long, lateral, surface roots with well-developed tap or sinker roots to access ground water (Canadell et al., 1996; Gile et al., 1998; Scott et al., 2004; Grigg et al., 2010). Deeply-penetrating roots can facilitate infiltration of water and dissolved organic carbon (DOC) to greater depths (Archer et al., 2002; Rumpel and Kögel-Knaber, 2010), and establish microbial communities and SOC fluxes there. Deep-set SOC can comprise at least half the total SOC (Harrison et al., 2011); it accumulates and is destabilised through a range of processes, and generally has a greater average age than that in surface horizons (<0.2 m), e.g. in the order of a thousand years (Rumpel and Kögel-Knaber, 2010). Therefore it is important to conceptualize its affect when designing rangeland SOC experiments. Few experiments have analysed the contribution to SOC from coarse roots (Rumpel and Kögel-Knaber, 2010) but there is increasing recognition of their contribution (e.g. Hancock et al., 2007; King et al., 2007; Kirschbaum et al., 2008; Nygren et al., 2009). Coarser woody-roots decompose more slowly (Ludovici et al., 2002; Kemp et al., 2003; Janisch et al., 2005). With the longer timelines involved in deep-set C and coarse woody-roots, measurement of SOC over a large extent of the t-axis is more critical, when there is change in woody vegetation.

8.2.3 Mechanisms in this chapter

Following the introduction to SOC distribution in rangelands given above, the relative benefits of one-, two- and three-pool SOC models is demonstrated, then one of those is used to emulate results from the CENTURY model (a three-SOC-pool system with feedbacks Parton et al. (1987)). CENTURY was not run directly

because: (a) the amount of site-specific data required (e.g. for climate, plant growth and decomposition) was could not be obtained within the confines of the project, (b) the number of sites examined was onerous, and (c) I wanted to display the keystones of the dynamics transparently, by using the simplest, applicable mathematics.

In the first example the influence of the time axis (t-axis) and z-axis (depth) in data sampling, processing and interpretation is addressed. Rangelands in the southwest USA were overgrazed in the 19th century, inducing degradation, with erosion, net C efflux, and reduction in ecosystem function, followed by recovery and woody thickening (Lund, 2007; Wheeler et al., 2007; Wilcox et al., 2008). Sequestration of SOC associated with such thickening in a Texan grassland was found to approach an asymptote with a nearly fourfold increase in SOC after ~2,500 years (Hibbard et al., 2003). Conversely, in Jackson et al. (Jackson et al., 2002) where rainfall was above ~460 mm year^{-1}, thickening of mesquite (*Prosopis* spp.) was found to induce net SOC emission. This latter study was unique in that it tested for roots nearly as far down as the lowest root of a phreatophyte (in this instance to 10 m), and proportional change in SOC with thickening, was determined to 3 m depth. Its principal conclusions were that woody thickening incurs a net emission in wetter sites and that 'assessments relying on C stored from woody plant invasions to balance emissions may therefore be incorrect'. The findings, published in the prominent journal 'Nature', caused apprehension over whether or not the USA national C sink had been overestimated (Goodale and Davidson, 2002). While scepticism was voiced, there were no compelling arguments (Archer et al., 2004; Wheeler et al., 2007; Liao et al., 2008) and controversy remains.

In the second example data interpretation on the time axis and the x-y plane is scrutinised. Approximately 22 Mha of rangeland in Australia has been deforested for intensified, commercial grazing (Chapter 9). Curtailing the accompanying change in biomass was instrumental in Australia's adoption of the Kyoto Protocol, though the accompanying ΔSOC has not yet been sufficiently reconciled (Henry et al., 2002). Rates for the State of Queensland (QLD), Australia, have been the highest nationally for several decades, e.g. ~0.343 Mha year^{-1} in 2006, 59% of which was remnant

native vegetation (DNRW, 2008). Data from a comprehensive study of deforestation in QLD is analysed and reinterpreted, where ΔSOC following deforestation ranged from –5.4% to 1 m depth (Harms and Dalal, 2003; Harms et al., 2005), to –31% for 0.05 m depth (Dalal et al., 2005). The former measurement contributed to high uncertainty for grazing businesses in a study considering GHG benefits of land rehabilitation (Bray and Golden, 2008).

Lastly, the way in which identification of the components of events and their C fluxes can facilitate concord between existing reports on ΔSOC for rangelands is discussed.

8.3 Methods

8.3.1 Formularizing the temporal dependence of ΔSOC

Insights into the basic temporal dynamics of ΔSOC emerge from the mathematical correspondence established between some ecological and radioactive processes (Jenny et al., 1949; Olson, 1959; Olson, 1963). The decomposition sequence for SOC, where C pools are linked in series, is conceptually similar to chain radioactive decay with the SOC stock equating to the radiation background. Principles from the mathematics for radioactivity can be applied to SOC. In aerobic conditions SOC formation from plant residues can be represented by a three-stage process, linked to particle size and time-dependent chemical or physical stability (Parton et al., 1987; Liao et al., 2006a; Grandy and Neff, 2008) or approximated by a 2-pool system (Trumbore, 2000; Kirschbaum, 2004), or by a one-pool system with a time-dependent decomposition rate (Yang and Janseem, 2000; Feng, 2009). The mathematics for SOC changes in such pool systems have been presented by Bruun et al. (2004) and in this section a system suitable for use in modelling the C dynamics of change in woody vegetation in rangelands is described.

Several principles arise for SOC from the comparison with chain radioactive decay:

1) the half-time of sequestration (i.e. half the time to reach maximum stock) is similar to the half-life of the slowest (longer half-life) pool,
2) the maximum stock is linearly proportional to the half-life of the slowest pool,
3) the decomposition rate is similar to that of the slowest pool, and
4) the time to accumulate 95% of maximum stock is triple the turnover time of the slowest pool (i.e. 3/decay constant) (Jenny et al., 1949).

These four principles apply directly to SOC sequestration under woody vegetation, with the following implications:

1) because the SOC arising from woody vegetation is more recalcitrant than that from herbaceous vegetation (due to its higher concentration of aliphatic root suberin, waxes, glycerides, lignin and tannins) (Liao et al., 2006a; Lorenz et al., 2007; Filley et al., 2008), then together with its deeper roots the time to 95% sequestration is longer and the resultant SOC stock is higher, and
2) because SOC longevity increases with depth (Dalal et al., 2005; Rumpel and Kögel-Knaber, 2010), the half-time for sequestration for the full soil profile is longer than that of the upper horizons alone.

Considering a one-pool model (Yang and Janseem, 2000; Feng, 2009), the amount of SOC remaining at time t (in years) is SOC_t, given by:

$$SOC_t = SOC_0 \exp\left(-(kt)^{\alpha}\right) = SOC_0 \exp\left(-k't^{\alpha}\right) \qquad \text{Eq8-2}$$

where SOC_0 is the initial stock, k' is equal to k^{α}, with k=2.39 and α=0.22, i.e. k'=1.21. After a short period this system becomes equivalent to first-order decomposition with a half-life of ~35 years. For example, SOC_{12} is essentially the same as with α=1 and k'=0.01959. Such rapid decomposition does not allow SOC to accumulate if shrubs or trees die more than a few decades apart. This absence of a "long tail" has been noted previously for a 1-pool model (Manzoni et al., 2009). The parameter values in Eq8-2 were derived from experiments mostly observing decomposition for less than 11 years (Yang and Janseem, 2000; Feng, 2009). Also,

most samples were of the more-labile carbohydrates, or were derived from arable-crop, herbaceous species. Such short half-lives, and absence of more stable compounds, are inapplicable to comprehensive rangeland SOC dynamics, which include subsoil and woody vegetation.

Manzoni et al. (2009) found that a 2-pool, or more complex systems, may represent longer-term dynamics better than a one-pool model. Consequently, parameters were chosen from rangeland studies that included woody vegetation and multiple pools.

Under woody thickening, a 3-pool system yielded SOC half-lives of 5, 300 and 393 years (Neff et al., 2009). Another thickening study (Liao et al., 2006a) yielded half-lives of 21, 42 and 250 years plus half-lives up to 770 years. The times to 95% sequestration (3/*k* of the slowest pool) for these two studies were 1,700 and 1,082 years respectively. These times to 95% sequestration are of the same order of magnitude as the ~2000 years from CENTURY modelling of mesquite thickening (Hibbard et al., 2003), to 0.2 m depth (a depth constraint on that version of CENTURY). Both 3-pool studies were within the upper 0.15 m of soil, therefore the half-lives and mean residence times (MRTs, or turnover times =1/*k*) for the full soil profile would be longer.

To encompass these two studies, ΔSOC is examined in the present work with: (a) a 3-pool system, with half-lives of 5, 150 and 500 years, and (b) a 2-pool system, with half-lives of 2 and 700 years (Figures 8.2). The SOC pools were interconnected and without feedback. Input to the SOC pool was from biomass decomposition following (a) continuous (annual) and (b) episodic biomass death (namely once every 200 years— bicentennially).

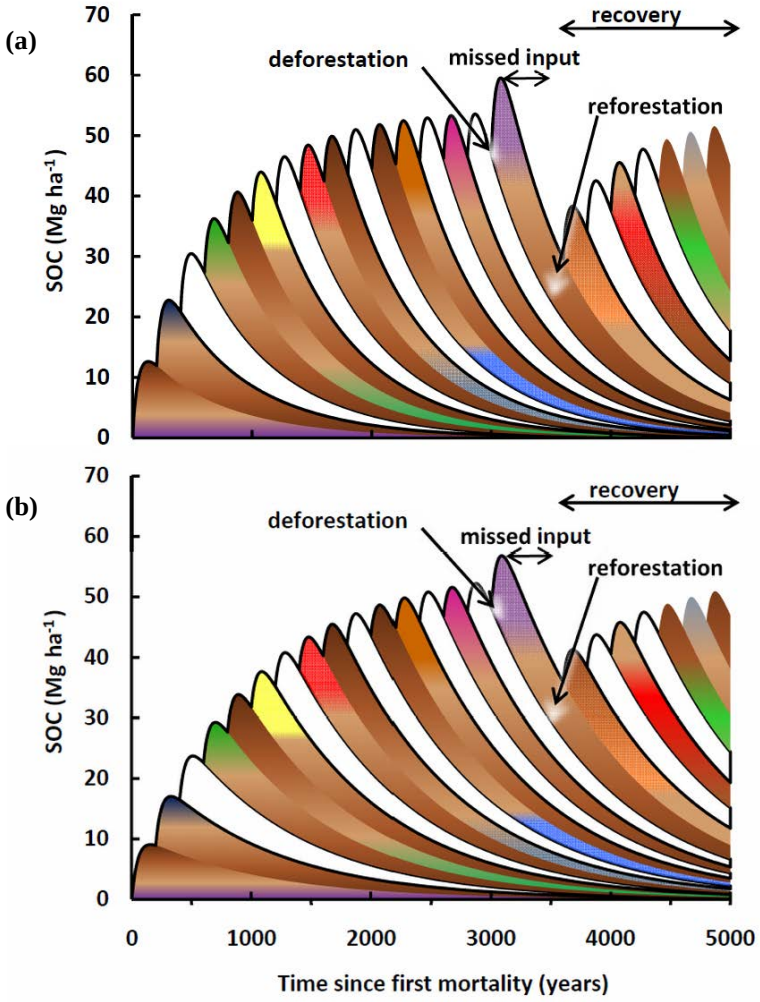


Figure 8-2. SOC sequestration over the full soil profile, accompanying maturation and concomitant decomposition of woody thickening. For **(a)** 3-pool and **(b)** 2-pool, systems with bicentennial input to SOC from tree mortality. Shades distinguish different biomass death cohorts. The SOC decomposition and net sequestration profiles are steeper for the 3-pool system. Deforestation incurs an initial peak before a decline, then a long duration for recovery after reforestation. For clarity the SOC from grass has been omitted from the deforestation epoch (but is included in Figure 8-4).

Biomass decomposition was modelled as exponential decay, with half forming nascent SOC and half emitted to the atmosphere. Of the nascent SOC, 55% was delivered to the fast pool, and 45% to the slower pool. Then 60% of the SOC from the fast pool went to the slow pool, along with 40% emission to the atmosphere (i.e.

'respiration'). This is similar to the 50:50 allocation for the 2-pool system in Manzoni et al. (2009). Similarly, in the 3-pool system, 60% went from the slow pool to the slowest pool, with 40% emission to the atmosphere. The slower-two pools in the 3-pool model were in series.

To achieve the same, long-term SOC stock of 52 Mg ha^{-1} for the two systems, inputs of C in dead biomass were 95 and 64 Mg bicentennially, for the 3- and 2-pool systems respectively. The higher influx rate needed for the 3-pool system was due to extra respiration, being from the intermediate pool, giving faster SOC decomposition of each pulse from decomposed biomass.

The difference between continuous and episodic input to SOC from maturation of thickening was compared for both the 2- and 3-pool systems (Figure 8-2). Episodic input, such as from mortality accompanying self-thinning, senescence or fire produces undulating net SOC; whereas continuous (annual) input, e.g. representing litter fall, fine root turnover, exudates, or slow senescence, produces a smoother curve. The total SOC flux and maximum stock are approximately identical whether the input is bicentennial or annual, so long as the time-averaged input rate to the soil is the same. In reality, however, if the annual deposits are derived from fine roots, then they may contain substances with different decomposition rates than from the more-episodic input from coarse-root death, and thereby produce different SOC stocks. There are few reports of relative contributions to the total, from continuous turnover and from episodic mortality, but one estimate (Chestnut et al., 1999) suggests they are approximately equal.

The annual input to SOC produced a logistic-shaped curve for ΔSOC, which could be approximated by exponential curves prior to, and after, the inflection point (Figure 8-3). This concurs with other modelling of ΔSOC (Wynn and Bird, 2008), and mathematical approximations for logistic curves (Birch, 1999; Ford, 1999; Flynn and Ford, 2005). The use of logistic curves for modelling forest biomass growth is well-established (e.g. Bruce and Schumacher, 1942; Knuchel, 1953). The inflection points for the 2- and 3-pool models were 154 and 130 years respectively. Prior to the

inflection point on the logistic curve, is the early stage of thickening, where juvenile stands self-thin, and where therefore input to SOC from biomass decomposition is likely to be more continuous than episodic. The long sections of the curves along the t-axis could be approximated by exponential functions, which allowed calculation of sequestration timelines assuming first-order kinetics.

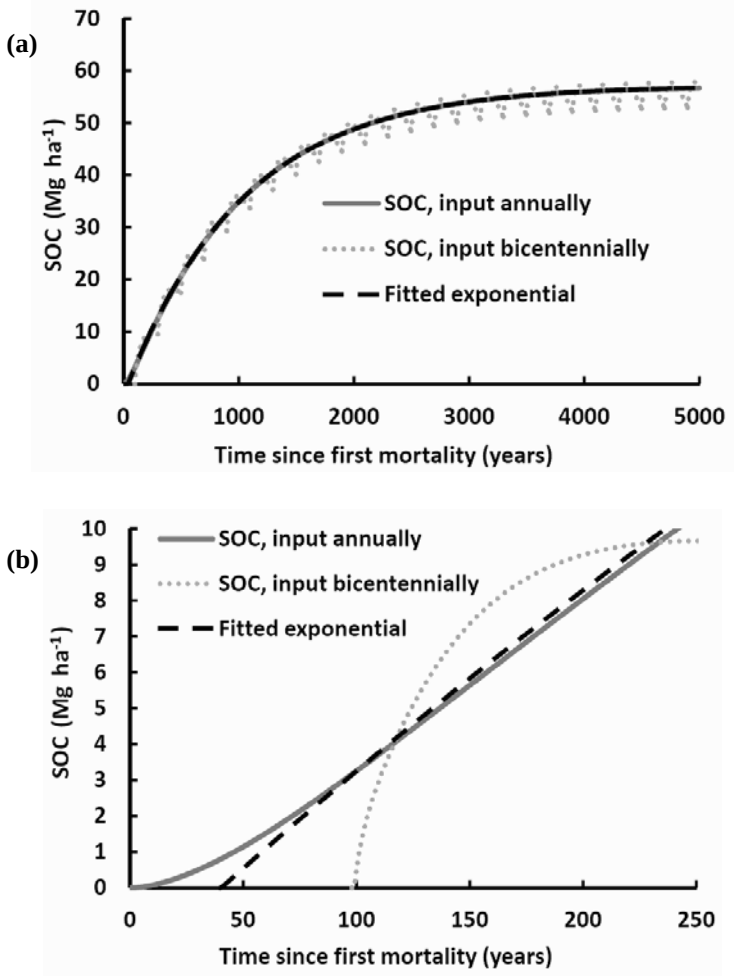


Figure 8-3. SOC sequestration for woody thickening; 2-pool model. Biomass mortality supplying input to SOC and wood decomposition is simulated by two separate modes: episodic— 50 Mg ha^{-1} once every 200 years (undulating, grey, dashed line); and continuous— 0.25 Mg ha^{-1} annually (solid grey line). Average, long-term sequestration trends are the same for both supply rates. The curve fitted by regression (black, dashed line) to the continuous input shows that long-term sequestration was approximately exponential **(a)**, but close up of the first century shows it is actually logistic **(b)**.

The similarity in sequestration curves for the 2- and 3-pool systems (Figure 8-2) meant that a 2-pool system could be used for basic visualisation of changes in relative proportions of grass- and woody-cover. Long-term SOC stocks and fluxes, for grassland and thickening were adapted from CENTURY output from Hibbard et al. (2003). The 2-pool model was fitted to that data to show diagrammatically the effects of a sequence of major impacts on SOC: land degradation, thickening, deforestation and reforestation (Figure 8-4). For woody growth the input to SOC was modelled as 50:50 annual and bicentennial; input from grass was annual. Deforestation was simulated simply by an extra pulse of dead biomass (corresponding to the freshly killed trees) followed by stopping input to SOC from woody vegetation for 200 years and meanwhile generating annual input to SOC from grass. Reforestation (with reduction in grass cover) was simulated by re-starting the input to SOC from woody thickening and stopping the input from grass.

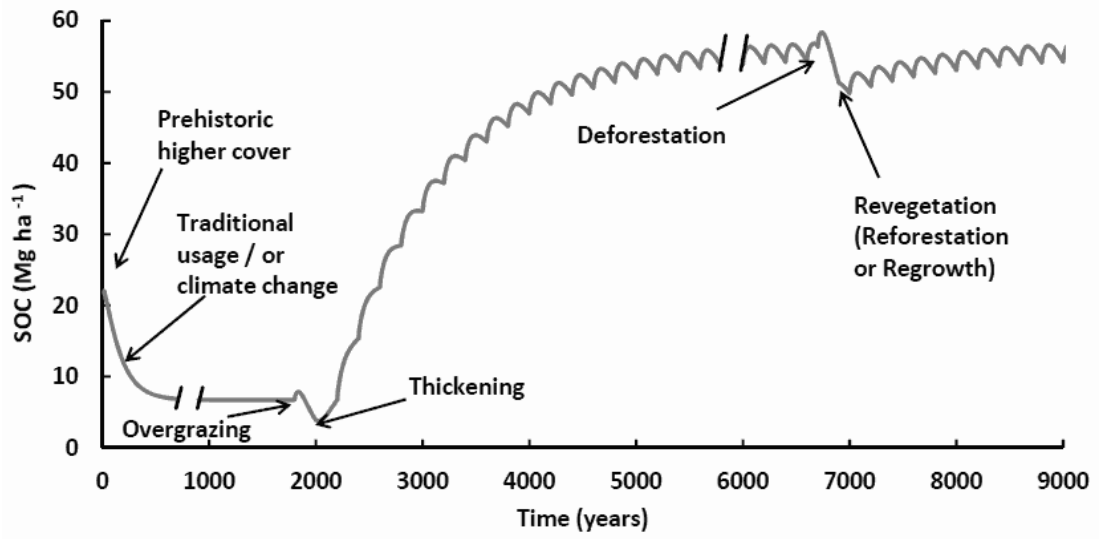


Figure 8-4. Time series of ΔSOC accompanying thickening, deforestation and reforestation, from a 2-pool model for SOC. The rippled line, corresponding to ΔSOC from woody growth, results from half-annual and half-bicentennial turnover. Chronological events: traditional usage or climate change reduces vegetative cover to a savannah or grassland, subsequent overgrazing and land degradation reduces vegetation cover and loses topsoil, thickening incurs sequestration, deforestation (plus new grass growth) incurs C emission, and revegetation incurs sequestration. Note long timelines involved

after management influences, before substantial change occurs in SOC, e.g. for reforestation following an absence of woody vegetation for 200 years.

To show how the 2- and 3-pool models represent the age demographics of SOC, the fraction of SOC present from each age cohort (i.e. from time since biomass death) was graphed as a function of time (Figure 8-5). The creation date for SOC was selected as when nascent SOC production was highest from each tree-mortality episode. At any instance in time there was exponentially more SOC from the more recent cohorts— concurring with Bruun et al. (2004). The average SOC age for the 2- and 3-pool systems were: 1,005 and 732 years respectively. Error margin in these figures resulted from the gradual release of dead biomass to SOC. The resulting average ages for SOC are close to the MRTs for the slowest SOC pools (1,010 and 721 years for the 2- and 3- pool systems, respectively), matching expectations from first-order kinetics (Manzoni et al., 2009). Thus the simple models adequately represent the major age demographics of SOC.

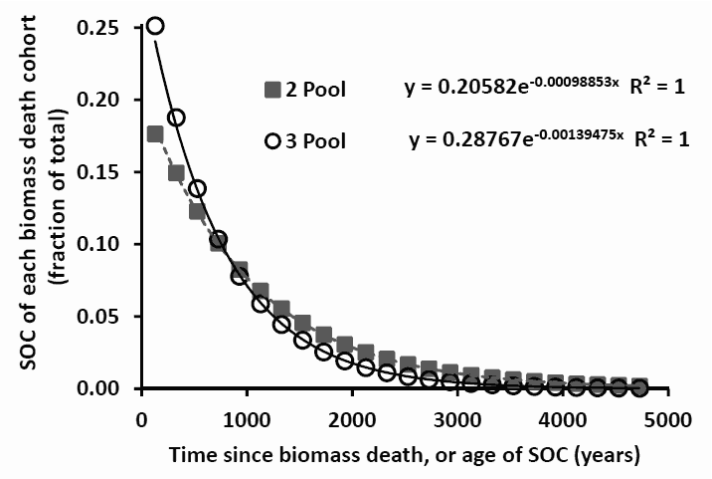


Figure 8-5. Proportions of SOC present, as a function of when the contributing biomass died (i.e. age of SOC). For this simulation the average times since biomass died was 732 and 1005 years for the 3-pool and 2-pool systems respectively; i.e. close to the turnover times of the slowest pools. Note that at any one instance in time there is a range of ages of SOC, with more of the younger SOC.

Following the logistic curve shapes (Figure 8-5) for the net sequestration effects of SOC in the 2 and 3-pool models above, and for SOC in grassland degradation and woody thickening (Hibbard et al., 2003), it was appropriate to employ logistic curves to display the net effects of land degradation and thickening combined. The form of logistic function used to represent SOC_t from grassland and thickening was:

$$SOC_t = SOC_0 + flux\_mag\left(1 - \frac{1}{1 + \left(m \times t^k\right)}\right) \qquad \text{Eq8-3}$$

where: t is time, SOC_0 is the initial stock, *flux_mag* is the maximum change in stock (>0 for sequestration and <0 for emission), and m and k are fitted parameters.

8.3.2 Example 1— Time and depth axes

To simulate SOC(t) for grassland degradation and thickening the events described in the thickening study of Jackson et al. (2002) were modelled using Eq8-3. An approximate onset date for degradation was 1850 (Hibbard et al., 2003). The time since onset of thickening equals the stand age. To parameterize the SOC levels in Eq8-3 data were fitted from the output of CENTURY modelling by Hibbard et al. (2003), as there was insufficient information in Jackson et al. (2002) for parameterisation (e.g. no asymptotic stocks). Thickening was modelled as beginning at two possible dates: in 1850 (scenario 1, simultaneously with onset of land degradation), and in 1949 (scenario 2, 99 years after establishment of degradation) (Fig. 6b). The onset date for scenario 2 was chosen from the average stand age in Jackson et al. (2002), which concords with a SOC isotopic study of mesquite thickening in Texas (Boutton et al., 1998) which found that thickening began less than 100 years ago. That latter study also found that thickening started at different times on different topographic landscape components (which validates testing more than one date for onset of thickening). However, in their study there was the additional complication of SOC translocation together with different stand ages in different landscape positions, which prevented further interpretation of SOC age.

With time being crucial to the sign of ΔSOC the original data from Jackson et al. (2002) were re-aggregated as a function of stand age.

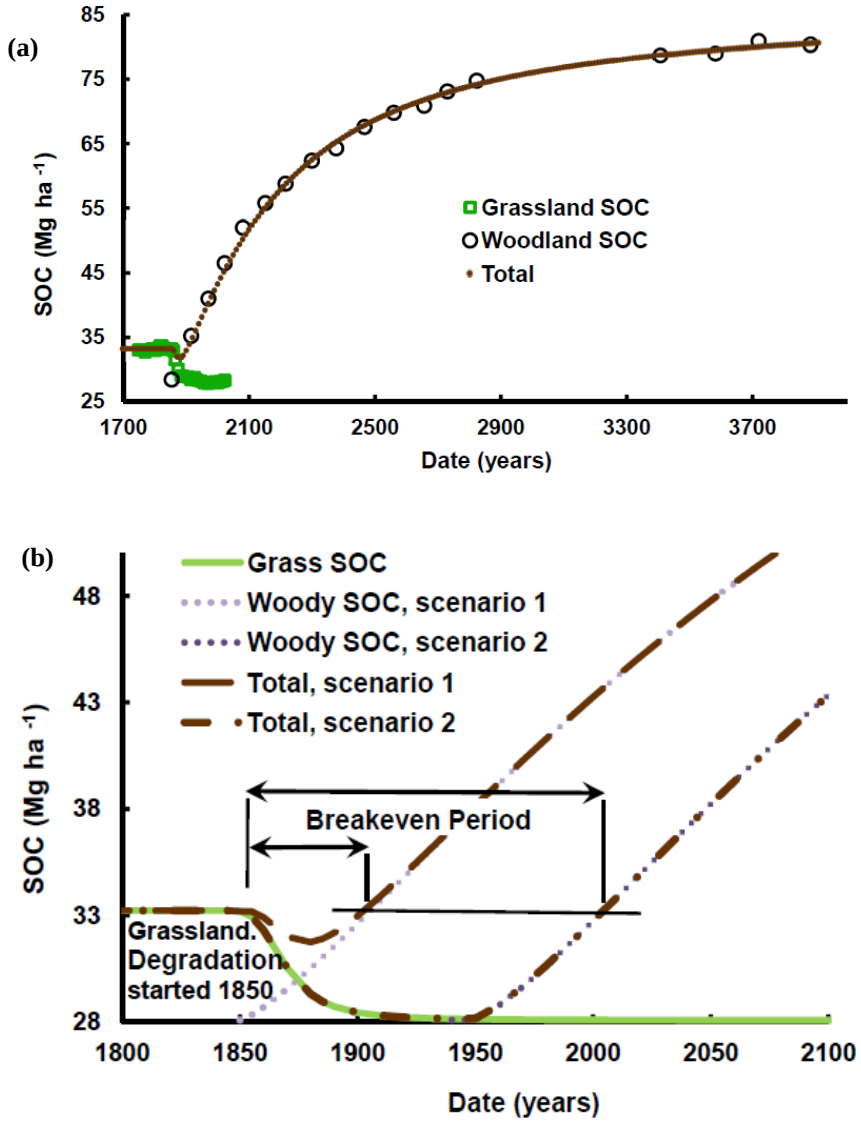


Figure 8-6. (a) Modelling of SOC (to 0.2 m depth) accompanying grassland degradation and mesquite woody thickening. Data points are from CENTURY modelling by Hibbard et al. (2003). Thickening starts simultaneously with land degradation, in 1850. (b) Close-up of (a) near the onset of thickening, but with two different scenarios for when thickening begins. Totals for two thickening scenarios: 1—thickening starts in 1850; 2—thickening starts in 1880. Straight lines show the breakeven period—the time taken to regain SOC to the pre-degraded grassland level. Measurement of ΔSOC before and after the breakeven date will record emission and sequestration respectively.

Each of the six data points in Jackson et al. (2002) consisted of a pair, with one member ostensibly corresponding to un-degraded grassland and the other member corresponding to a woody thickened site, with other factors being equal within each pair. Data were checked to see if they complied with that pairing condition. From Google-Earth Inc. imagery, it was observed that all study sites except Riesel were in extensive land-use zones, whereas the Riesel site was in an intensive, arable agriculture zone, containing regenerated prairie (Potter et al., 2004). Possible implications of this are: (a) the thickened member of the paired site may have been too degraded for development as arable land (compared with neighbouring prairie), or (b) it had a locally unique substrate, or (c) the grassland member of the pair had received remedial treatment (artificially incurring sequestration). This suggests a complex site history that could be grounds for discarding the Riesel site data point from the SOC trend investigation. Site history influences C distribution on the z- and t-axes, not just on the x-y plane. Also the Riesel site had a different trend with depth for $^{87}Sr/^{86}Sr$ isotope ratios to the other sites, and a second peak in SOC starting at 9 m depth (the z-axis), which was higher for the thickened area; but this extra C was not included in the regression with rainfall in the original study (Jackson et al., 2002). These two factors gave grounds to doubt the validity of Riesel as a paired site, and consequently data were tested for correlations with and without that site.

8.3.3 Example 2— Time and horizontal axes

In this example of deforestation for grazing, using studies from QLD, representation on the t-axis was approached by grouping ΔSOC data as a function of time. Representation in the x-y plane was considered by: (a) calculating temporal trends for data from only environmentally-equivalent sites, and (b) spatial averaging of data.

In the original study (Harms et al., 2005) space was substituted for time by choosing sites deforested at different dates and by pairing those thought to have been environmentally equivalent prior to deforestation. Across 50 Mha there were 32

locations, represented by 47 paired sites. SOC was sampled to 1 m depth. Environmental differences (e.g. texture, including clay content; microbial species; moisture availability; temperature; and tannin and lignin contents of plant species) can strongly affect decomposition rates and amounts of biomass debris (Cornwell et al., 2008). To assist in controlling for such spatial heterogeneity, and to avoid bias from the early oscillations in fluxes (García-Olivia et al., 1994), only sites for which there were two or more distinct sampling dates after deforestation were used. One location, Kindon, had improperly paired sites owing to differences in clay content, cation exchange capacity and profile mixing (Harms et al., 2005) and was not used in the re-evaluation.

The percentage change (from the pre-deforestation SOC stock) was graphed as a function of time since deforestation. The original 32 locations became nine trend lines. These lines formed vectors, which together, revealed a flux field (a separate change vector for each location). The average SOC efflux, was 0.58(0.54) % year^{-1}. That average efflux rate was summed over the 5–35 year experimental time span to give the total SOC efflux for deforestation.

Insight into the time and environmental dependence of different spatial averaging methods can be provided by the [non-linear] formulation of SOC(t). Following deforestation, ΔSOC can be represented by a logistic curve which, after the initial fluctuations and prior to the inflexion point, can be approximated by an exponential function (see section 3.1):

$$Ctot = Cg + \left(\left(Cw - Cg\right) \times \exp\left(-kt\right)\right) \qquad \text{Eq8-4}$$

where t is the time since deforestation, $Ctot$ is the SOC after time t, Cw is the long-term-average SOC under woody vegetation, Cg is the long-term-average SOC under grass only, and k is the decomposition rate constant. The short-term oscillation owing to the sudden influx of dead roots (Figure 8-2) is ignored here. Where deforestation incurs sequestration, k is negative.

Spatial averages for ΔSOC with deforestation can be the average of all the percentage changes (the "disaggregated average", DA, giving ΔSOC_{DA}), or the percent change after aggregation of SOC stocks (the "aggregated average", AA, giving ΔSOC_{AA}). For the ratio of ΔSOC between the grassland and woody states, Jensen's inequality (see section 8.2) states that $\Delta SOC_{DA} \neq \Delta SOC_{AA}$, i.e.:

$$\frac{1}{n}\sum_{j=1,n}^{j}\left(Ctot_j - Cg_j \Big/ Cw_j\right) \neq \frac{1}{n}\sum_{j=1,n}^{j}\left(Ctot_j - Cg_j\right) \Big/ \frac{1}{n}\sum_{j=1,n}^{j}\left(Cw_j\right) \qquad \text{Eq8-5}$$

where *n* is the number of paired sites. Thus AA is mathematically incorrect when there is environmental variability, or where the values being measured are correlated to at least some degree and the function is non-linear, as is usual in the real world. However, if either C_g or C_w are constant across all sites (i.e. invariant in x-y), or if C_g is not correlated to C_w (i.e. if ΔSOC is not related to the initial stocks) then the inequality may vanish.

Effects of the different methods of spatial averaging, along with different, speculative decay constants for two example sites were examined. The AA was used in Harms et al. (2005); therefore the original data were re-processed using DA.

8.4 Results

8.4.1 Example 1— Time and depth axes

The results for land degradation followed by thickening (Fig. 6) were:

1. Recovery of SOC to pre-degradation level (i.e. 'breakeven') was attained at dates 1903 and 2004, for scenarios 1 and 2 respectively. That is, the breakeven periods were 53 and 154 years respectively, and the stand ages at the breakeven date were 47 and 55 years respectively.

2. Emission was greater for scenario 2, i.e. when onset of thickening was delayed (15% (5.0 Mg ha^{-1}) compared with 4% (1.25 Mg ha^{-1}), in this example).
3. If measurements were taken either side of the breakeven date then they would provide a different sign for ΔSOC, i.e. they would yield contrary conclusions. Measurements prior to breakeven would indicate emission; close to breakeven would yield ΔSOC=0; and after breakeven— sequestration. Only the Riesel site showed emission after the breakeven period, and there was ample reason to disregard that site. In Jackson et al. (2002) sites were sampled on both sides of the approximate breakeven date. More generally: short-term ΔSOC experiments, with durations near the breakeven date can give contrary results.

4. If the minimum measurable ΔSOC is 20% (e.g. Huntington et al., 1998; Yanai et al., 2003b), then no significant change would be recorded until 1967 and 2065, for scenarios 1 and 2 respectively, i.e. ~110 years after onset of thickening. No stands in Jackson et al. (2002) were that old.

5. Breakeven would occur earlier if degradation ceased after onset of thickening.

The number of samples required to discern change is proportional to an inverse power of the time since change began (Poussart et al., 2004). Thus for a 90% chance of a discernible ΔSOC, ~75x more samples are required if measured after 5 years rather than 50 years since change began (Poussart et al., 2004). Only half the data points in Jackson et al. (2002) had a stand age of 50 years or older— indicating that more samples may have been necessary when measuring the younger stands.

With elimination of the Riesel site there appeared a weak, positive correlation between ΔSOC and stand age, R^2= 0.39, *P*= 0.1, N= 5, which was absent when the Riesel data were included (Figure 8-7). From the line of best fit it appeared that ΔSOC=0 when the stand age was 48(±10) years. That value is similar to the stand ages at the breakeven dates for scenarios 1 and 2, and those reported elsewhere for

thickening in southwest USA (Hughes et al., 2006; Wheeler et al., 2007; Throop and Archer, 2008). It is likely that a more definitive trend between ΔSOC and stand age could be achieved with appropriately purposeful experimentation.

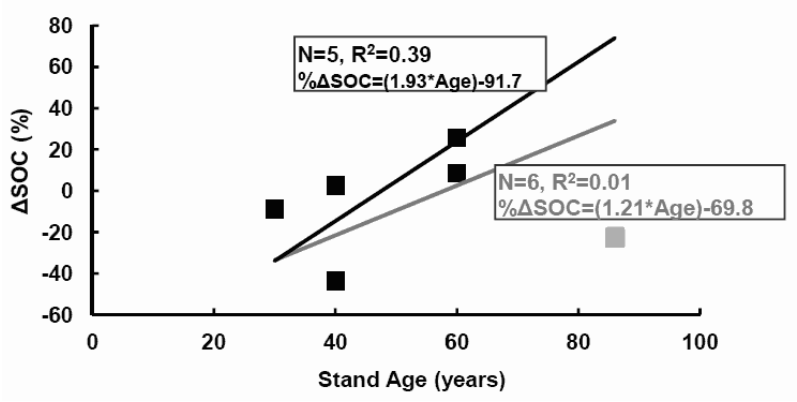


Figure 8-7. Data from Jackson et al. (2002) re-aggregated as a function of stand age. For calculation of the straight line of best fit, the standardised major-axis analysis method (Warton et al., 2006) was used as uncertainty existed in both abscissa and ordinate. Lines of best fit: grey—all six data points; black—all data points minus that for the Riesel site (grey point). Note that a weak trend for ΔSOC(t) appears with elimination of the Riesel data.

The modelling suggests that sequestration may be significant for a few more centuries after the breakeven date. This indicates that much longer periods of measurement are required (i.e. a longer span of the t-axis) to make conclusions on the direction of ΔSOC with thickening, above the average stand age of 53 years used in Jackson et al. (2002). Also, this re-evaluation shows that site pairing is not just an issue in the x-y plane but site-based influences also arise from the z- and t-axes.

Note that elimination of the Riesel data did not significantly alter the relationship between ΔSOC and rainfall found in Jackson et al. (2002) as it was supported by the remaining five data points. Nor did the findings in the present work totally contradict the conclusion in Jackson et al. (2002) that GHG emission offsets may not be achieved through woody thickening— instead the findings confirmed that conclusion was true at the time of their experiment. For example, for scenario 2— breakeven was achieved in 2004 but not measurably so until 2065 (both dates being later than

the experiment by Jackson et al. (2002)). Furthermore, the parameterisation of the logistic curves was based on the 0.2 m depth work of Hibbard et al. (2003), and with deeper horizons having slower kinetics, i.e. requiring sampling of a comprehensive extent along the z-axis, then the breakeven date may be even later than that shown in Figure 8-6.b. Hence, the work here showed that the conclusion in Jackson et al. (2002) would require adjustment (examination of a longer period and sampling SOC to at least rooting depth) to account for the fact that although GHG emissions were not, at the time of measurement, offset by woody thickening in wetter areas, they may well be offset at some time in the future.

The 4D perspective indicates that the data in Jackson et al. (2002) fit ongoing sequestration, but it wasn't otherwise obvious because sampling was done over a small extent on the t-axis. With further time the reanalysis indicates that SOC under woody thickening is likely to increase, including in the higher rainfall regions. Consequently, the reanalysis portrays the data in Jackson et al. (2002) as being in concord with the main body of work on thickening, i.e. that thickening engenders net C sequestration.

8.4.2 Example 2— Time and horizontal axes

The SOC efflux for deforestation over the 5–35 year experimental time span, was 17%. That is higher than the 5.4% reported in Harms et al. (2005) where the spatially- and temporally-averaged absolute changes were used. Note that although ΔSOC was calculated as % $year^{-1}$ here (averaged over a specific time period), the rate is not constant and after a longer time period will approach an asymptote.

Additionally, Figure 8-8 indicated that emission will most likely continue beyond the time period of the original study (Harms et al., 2005). This indicates that the ΔSOC corresponding to the asymptotic situation may be higher in magnitude than –17%.

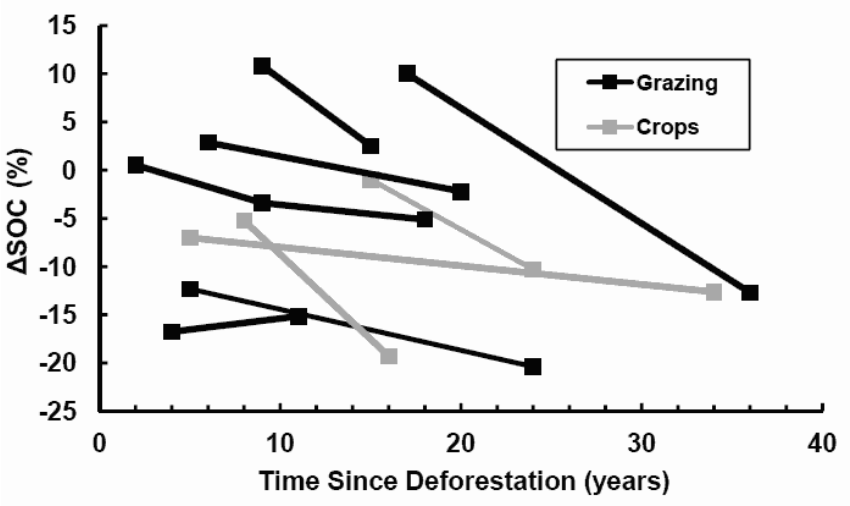


Figure 8-8. Chronosequence trends in SOC after deforestation, as a function of time since deforestation for sites in QLD (original data from Harms et al. (2005)). Percentage change is compared to the pre-deforestation SOC stock. Points joined by lines correspond to paired sites. After initial perturbation with deforestation, emission continues for at least several decades. The efflux rate is the averaged slope of the lines.

Data for two example deforestation sites from Harms et al. (2005), re-processed using DA, are presented in Table 8-1 and the corresponding SOC(t) is shown in Figure 8-9. The DA for those sites varied more strongly with time since deforestation than did the AA, and generally DA showed a stronger influence on %ΔSOC than did AA (Figure 8-10), which must therefore be accommodated in data sampling and analysis.

Table 8-1. Data for two sites deforested for grazing in QLD. Cw is SOC of the woody state and Cg is SOC of the grassland state. k is the rate constant of exponential decomposition (Eq8-4). Corresponding ΔSOC with time is in Figure 8-9. Species: *Acacia harpophylla* (brigalow), *A. aneura* (mulga) and *Eucalyptus melanophloia* (ironbark).

	Soil	Woody species	Date deforested	*Cw* (Mg ha^{-1})	*k* (year^{-1})	*Cg* (Mg ha^{-1})
Site 1	Vertosol	*A. harpophylla*	1972	168.6	0.004301	16.86
Site 2	Kandasol	*A. aneura* and *E. melanophloia*	1992	42.94	0.01031	4.294

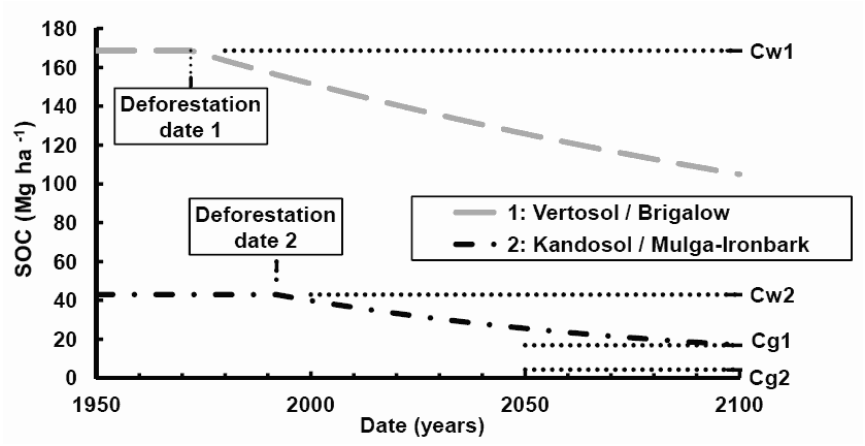


Figure 8-9. Initial SOC stock and change in time for deforestation of two example sites in QLD. Percentage change for two sites, deforested at different times. 1: high SOC— Vertosol with brigalow or gidgee (*Acacia cambagei*), and 2: low SOC— Sodosol with Ironbark. In site (1) deforestation occurred 40 years earlier. Corresponding data in Table 8-1.

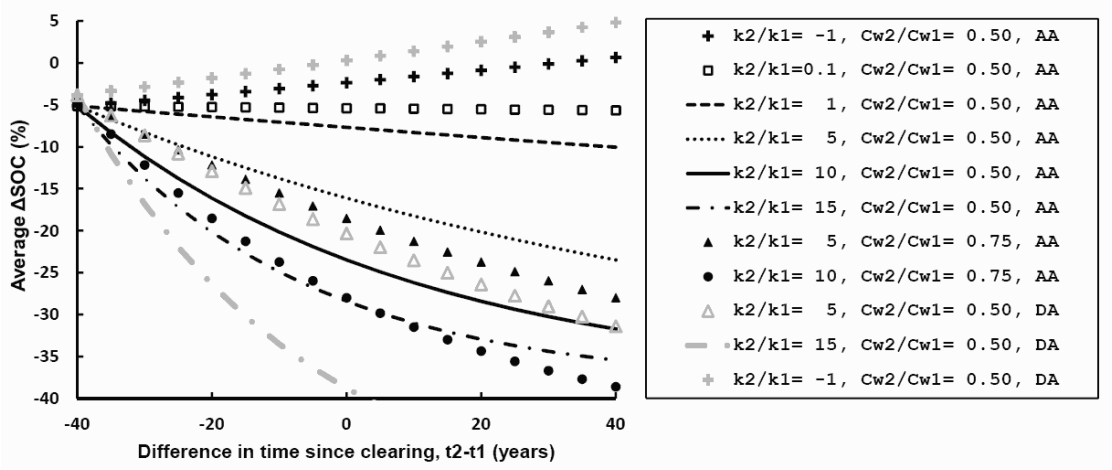


Figure 8-10. Simulated examples of ΔSOC (%) with deforestation, from different spatial averaging methods, different levels of woody-phase SOC, and different decomposition rates (*k1* and *k2*) (grey: disaggregated averaging, DA; and black: aggregated averaging, AA). Note that there are strong effects on the spatial averages of ΔSOC data resulting from: (a) the relative initial SOC stocks, (b) the time since deforestation, and (c) different decomposition rates; — presenting implications for data collection and analysis.

8.5 Discussion

8.5.1 Implications of time dependency

The similarity between half-life of the slowest SOC pool, and the half-time for sequestration of SOC, reaffirms that long time spans are necessary for ΔSOC experiments. The 4D conceptualisation of biomass distribution and decomposition (Figure 8-1) demonstrated that the representative SOC over 3D, at any one instance in time, is dependent on the historical and spatial distribution of tree mortality, and combined with the temporal conceptualisation (Figure 8-2), reveals how SOC approaches its asymptote after multiple stand generations.

Apart from the timelines in the mathematical modelling presented here, other factors suggesting that deforestation studies require long extents on the t-axis are: (a) the short-term peak in SOC due to decomposition of the nascent root and surface debris (Figures 8-2, 8-4); (b) the short-term direction of SOC flux varies with location (the x-y plane) (Harms et al., 2005); (c) for native pastures the positive impact of deforestation on productivity may be quickly followed by reductions in ecologically beneficial soil properties (Kaur et al., 2007), which would lead to C emission in the long-term; and (d) extra emission may occur through the 'priming effect' (Fontaine et al., 2007)— when substrate limitations on microbial mineralisation of the more passive SOC are released by a fresh supply of more-easily assimilated root compounds [from nascent root litter]. Deforestation studies also require long extents on the z-axis, especially where woody vegetation may access deep-set ground water.

Different models can yield different trends for age distributions of SOC (Bruun et al., 2004)— with more complex models forecasting greater longevity of some SOC pools (Manzoni et al., 2009). Results in the present work showed some slight

differences between the 2- and 3-pool systems in the shape of their sequestration and emission curves, and SOC age demographics. The logistic functions used here (e.g. to determine the breakeven period) modelled the output from a complex model (CENTURY, which includes for example, feedback between pools), and therefore the modelling of thickening (example 1) inherently included effects from such complexity. Possible long-term fluxes, beyond the time-span of measurement, were provided diagrammatically by the 2- and 3-pool models but modelling of those sites with the latest version of CENTURY could yield a more accurate forecast of long-term SOC flux.

8.5.2 Importance of 4D experimental integration

Both examples studied here showed the importance of controlling for multiple dependencies, which can operate on more than one axis and require extensive data collection. The use of mathematical models to determine the time span needed for ΔSOC experiments relies on existing information about the system. There is therefore a bootstrapping process (or 'power analysis') to measurement of ΔSOC.

Attention to representative sampling and data processing is applicable to all four axes. For example, both DA and AA are non-linear (and thus not interchangeable) with respect to the rate constant k (i.e. environmental variation in the x-y plane and also on the z-axis Dalal et al. (2005)) and time t (Eq8-4).

Influences of site history on SOC are interconnected in 4D, which must be accommodated, e.g.:

1. farmers deforesting the more productive (i.e. probably higher SOC) land first (Holmes et al., 2006) means that comparing C stocks in long-deforested land with those in more-recently deforested land (vestigial, less desirable), is inequitable;

2. although effects of thickening and land degradation may both be apparent simultaneously on the t-axis (e.g. Figure 8-11), the site history (which often provides clues to degradation) must be considered; and
3. substituting space-for-time can be inferior to long-term experiments, due to spatial heterogeneity, the likelihood of improperly paired sites (Harms and Dalal, 2003), and interconnections for SOC in 4D.

Figure 8-11. Degradation with concomitant thickening (southern ironwood, *Acacia estrophiolata* F. Muel., central Australia), showing that multiple effects can be concurrent, although not necessarily started simultaneously. The site has a long history of overgrazing; trees germinated after heavy rains following drought in the late 1960s, constituting wood thickening.

Longevity of SOC usually increases with depth due to, for example, apparent increased SOC recalcitrance, which may be due to substrate limitation, reduced decomposition activity, or specific mineral associations; rather than inherent differences in chemical structure of source material (Fontaine et al., 2007; Marschner et al., 2008). Also, in semiarid rangeland there is an effect from rainfall penetration with different pluvial duration— longer events (which are less frequent) allow more emission at greater depth whereas light rain only induces microbe-mediated emission at shallow depths (McLain et al., 2008). Longevity of SOC pools derived from roots varies not only with depth but between woody species and environments (Pansu et

al., 2009). In some soils, bioturbation may mix the age cohorts of some forms of SOC, even to 2 m depth (Pressenda et al., 1996). At wider scales the C budget can be affected by transport, during erosion and groundwater transit for example, and then by dynamics at the deposition site (Greene et al., 1994; Lal et al., 2004; Hoffman et al., 2008).

Due to physical difficulties of measuring SOC under tree trunks or coarse lateral roots, sometimes bulk excavation must be employed rather than coring, for both coarse root and SOC sampling (Archer et al., 2002; Resh et al., 2003; King et al., 2007). It is unclear if near-stem regions were measured by Jackson et al. (2002) but coarse root biomass was not measured— if the reason for its omission was physical difficulty then that may have precluded measurement of the spatially representative SOC of thickening (*cf* Chapters 5 and 6). The meta-analysis of thickening data in Jackson et al. (2002) also showed a negative correlation for ΔSOC with rainfall. However, the measurement depth there was only 1 m, whereas thickening species can manifest SOC to at least 2 m depth (as demonstrated in the empirical section of that study)— i.e. a z-axis dependence. Furthermore, a broader meta-analysis showed that SOC was consistently increased through woody thickening (Eldridge et al., 2011a).

8.5.3 Wider implications

Lowering the carbon footprint of rangeland activities represents ecological engineering, namely for climate change mitigation. From the long timescales needed for ΔSOC determination there are wide implications for monitoring and verification of ΔSOC in rangeland rehabilitation projects. The assessment interval of 5 years in Kyoto verification is known to be unmanageable for ΔSOC determination for mesic, temperate land uses (Smith, 2004); and the present work shows that for some common events in rangelands, 30–90 years is too brief. This conclusion concurs with that of Piñeiro et al. (2006) who found grazing impact on ΔSOC to be definitive over several centuries but different in magnitude and direction at a decadal time-scale.

There is, thus, a need for proxies such as modelling or substituting space-for-time, although well-paired sites (with equal capacity for production) are often hard to find.

Arid land rehabilitation, can incur measurable SOC sequestration to 0.2 m after three decades under irrigated woody cover (e.g. Su et al., 2007). The carbon footprint of that activity is possibly borderline in the short term but after longer durations and deeper sampling it may well represent climate change mitigation. Tropical rainforest has been degraded to 'derived savannah', with ongoing degradation and biomass loss. Rehabilitation of such agricultural zones is aided by reforestation (Badejo, 1998). Modelling, such as that used here, would be necessary for speculative carbon credits which could add impetus to such projects.

Using a similar approach to that in the examples above, the discord in the literature over the sign of ΔSOC with rangeland deforestation for pasture can be re-examined. In reviews, the 4D breadth of the individual experiments is often unstated. One particular meta-review of ΔSOC with land-use change— Guo and Gifford (2002)— is often-cited (e.g. Goodale and Davidson, 2002; Grünzweig et al., 2007; Kirschbaum et al., 2008; Marin-Spiotta et al., 2009). One of its conclusions was that ΔSOC is negligible, or that sequestration is minimal, upon conversion from forest to pasture. Other reviews (McGrath et al., 2001; Murty et al., 2002) found ΔSOC= 0. This has serious implications for climate change mitigation when environmental management decisions (such as conversion of forest to pasture) hinge on finances of carbon balance (e.g. Harper et al., 2007). For example the lower figure derived by Harms et al. (2005) (than derived here) contributed to high uncertainty in a study of GHG budgets for grazing businesses with land rehabilitation (Bray and Golden, 2008). In Guo and Gifford (2002) a net emission of only 8% of SOC upon deforestation was found by combining 170 observations from 19 publications. There appear to be several factors contributing to uncertainty in that value:

1. it used several reviews with an unknown number of samples, therefore, the relative weights applied to different 'observations' may have been inequitable

2. several input data were from one Amazonian region where sampling was ≤0.3 m, and where the chronosequence most likely included preferential, earlier deforestation of land with higher initial SOC
3. their Fig. 2 showed that the trend of finding sequestration with deforestation was stronger if measured <0.3 m rather than for the full profile, nevertheless they averaged all values
4. several of the sites had earlier undergone selective-logging or a crop phase (with fertilisation etc. and accompanying SOC fluxes) before conversion to pasture (thus artificially altering C stocks)
5. many pastures had sowing of exotic species, and intensive management, which makes comparison inequitable (as the forest hadn't received the same attention to productivity, and the carbon footprint of the extra activity was not included).

Thus, there were a wide variety of previous land-use histories, all with unaccounted-for but most likely ongoing influences on SOC (due to long timelines for ΔSOC). Such influences can lead to unwarranted conclusions about ΔSOC if they are not accommodated in experimental design and analysis. Only close re-examination of past experiments, or carefully taking into account 4D factors in future experiments, will produce results suitable for reconcilable climate change mitigation. This is pertinent to reforestation, where SOC takes ~2 millennia to recover after woody vegetation was absent for only ~200 years (Figures 8-2, 8-4).

Both overgrazing and deforestation incur an initial peak in SOC followed by a long emission, through their effect on biomass (e.g. Irvine et al., 2007; Kirschbaum et al., 2008). Specifically, overgrazing decreases root volume (Villamil et al., 2001), giving a pulse of dead biomass to the soil, but decreased inputs in the long-term, when biomass ceases to be replenished. Overgrazing of woody vegetation (e.g. erosion and gradual decline and loss of riparian vegetation) is slower and patchier than broad-scale deforestation, and land rehabilitation after overgrazing can be arduous (Christian and Perry, 1969). Therefore, the extent of the t-axis used in experiments studying ΔSOC with overgrazing (and land rehabilitation) should be longer than that

used for studying more spontaneous deforestation and reforestation. Similarly, with appropriate sampling in 4D it is possible that uncertainty in ΔSOC with change in land management may lessen in the more-xeric rangelands.

In summary, the process of applying comprehensive 4D sampling and analysis, for quantifying rangeland SOC fluxes, has important implications for presently or previously wooded rangelands; such as: higher magnitude fluxes must be accounted for and longer time-spans for unambiguous measurement are necessary. Earlier findings on thickening or deforestation appear no longer contradictory when one considers that they only presented views from different sections of the 4D axes, some being too small in span on at least one axis. Specifically, using fundamental mathematics it can be seen that although thickening will not necessarily increase total SOC stocks in all situations, there is as yet insufficient evidence to suggest that SOC will decrease in the long-term in higher-rainfall zones with thickening. The necessary experimental and analysis methods described here, and their implications for C budgets, can be applied to property-level, regional and national carbon accounting. Application of the suggested approach should generally decrease uncertainty and so attenuate the impasse on ΔSOC forecasts for wooded rangelands, subsequently engendering financial viability in rangeland C trading and more reliable national GHG accounts, and in turn provide for more serviceable climate change mitigation.

Chapter 9 Carbon management of commercial rangelands in Australia: major pools and fluxes

This chapter is adapted from a published paper: Dean et al. (2012c)

9.1 Abstract

The rates, magnitudes and uncertainties for major carbon (C) fluxes for rangelands due to commercial grazing and climate change in Australia are examined in this chapter. Total net C emission from biomass over 369 Mha of rangeland to-date was 0.73(±0.40) Pg, with 83% of that from the potentially forested 53% of the rangelands. A higher emission estimate is likely from a higher resolution analysis. The total change in soil organic carbon (ΔSOC) to-date was -0.16(±0.05) Pg. Carbon emissions from all rangeland pools considered are currently 32(±10) Tg year^{-1} — equivalent to 21(±6)% of Australia's Kyoto-Protocol annual greenhouse gas emissions. The ΔSOC from erosion and deforestation was -4.0(±1.6) Tg year^{-1} — less than annual emissions from livestock methane, or biomass attrition, however it will continue for several centuries. Apart from deforestation a foci of land degradation was riparian zones. Cessation of deforestation and onset of rehabilitation of degraded rangeland would allow SOC recovery. If extensive rehabilitation started in 2011 and erosion ceased in 2050 then a ΔSOC of -1.2(±0.5) Pg would be avoided. The fastest sequestration option was maturation of regrowth forest in Queensland with a C flux of 0.36(±0.18) Mg ha^{-1} year^{-1} in biomass across 22.7 Mha for the next 50 years; equivalent to ~50% of national inventory agriculture emissions (as of mid-2011); and long-term sequestration would be 0.79(±0.40) Pg. Due to change in water balance, temperature and accompanying fire and drought regimes from climate change, the forecast ΔSOC from the forested rangelands to 0.3 m depth was -1.8(0.6) Pg (i.e. 38(12)% of extant SOC stock) resulting from a change in biomass from

2000–2100. For improved management of rangeland carbon fluxes: (a) more information is needed on the location of land degradation, and the dynamics and spatial variation of the major carbon pools and fluxes; and (b) freer data transfer is needed between government departments, and to the scientific community.

9.2 Introduction

This study is centred on the Australian commercial rangelands. Their ongoing degradation emits CO_2, whereas less-degrading management would reduce those emissions; furthermore, reversing degradation could sequester C (Howden et al., 1991; McKeon et al., 1992; Glenn et al., 1993; Walker and Steffen, 1993; Henry et al., 2002; Hill et al., 2006).

The Australian State of QLD (which has the highest recent rates of deforestation), maintains the most comprehensive records in Australia of deforestation from LANDSAT remote-sensing (Queensland Government, 2016). Emissions from biomass due to deforestation over the last decade were ~11 Tg-C $year^{-1}$ (from ~0.375 Mha $year^{-1}$), and the deforestation rate is decreasing (DNRW, 2008). Emissions from deforestation in QLD, from all sources, are likely to remain approximately at 2006 levels (Raison et al., 2009).

This study examines (a) the magnitudes and timescales of major emissions and sequestration for the commercial rangelands of Australia with change of forest cover, and (b) the effect of climate change on SOC; and the options and hurdles in recognising, assessing and managing the larger carbon pools and fluxes in rangelands are illustrated. As a subset of the national rangelands, the commercial rangelands in QLD are examined, where management has the most practical capacity to reduce emissions.

9.3 Methods

9.3.1 Study region

The Australian rangeland zone covers 661 Mha (Donohue et al., 2005). Excluding from that zone, reserves and mesic arable land, the commercial livestock properties occupy ~369(±5) Mha (Figure 9-1). Subsections of that area were selected for examination of livestock watering points and forest regrowth. State and Territory cadastre maps, combined with federal land-use maps were used to determine the commercial grazing land in GIS compatible format. In all calculations carbon was considered to be 50% by weight of dry biomass.

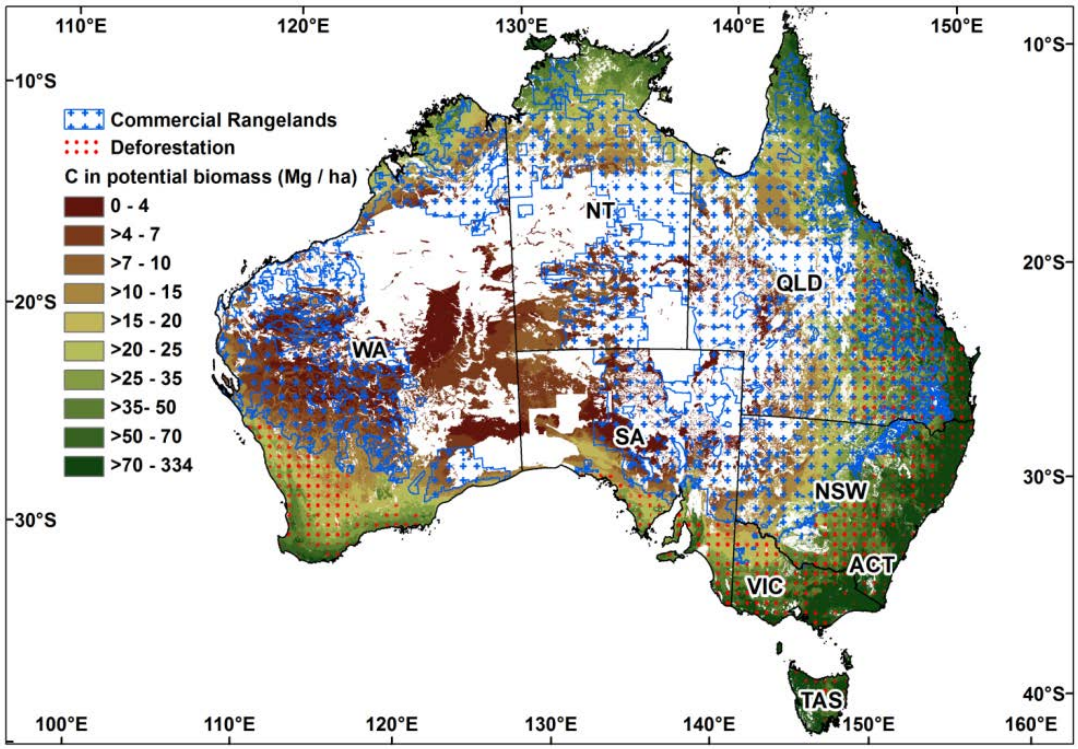


Figure 9-1. Distribution of C in potential forest biomass for potentially forested areas across Australia, showing intersection with the commercial rangelands and areas of deforestation.

Distance from water (for livestock) is the major influence on biomass, via reduced growth and dieback, with erosion elsewhere being the next major influence (Sparrow

et al., 2003). From qualitative observation of rangeland condition and associated carbon dynamics across Australia (Dean, 2005; Roxburgh et al., 2006a) it was noticed that watering points were frequently close to areas of higher biomass, and that the majority of those were riparian zones. Consequently the proximity of creeks and watering points were examined; the most readily accessible data was for ~6 Mha centred at -22.98º S, 135.27º E in the southern Northern Territory.

9.3.2 Nationwide emission/sequestration

9.3.2.1 Net ΔC to-date

An estimate of the total C emitted from biomass to-date due to commercial grazing, was obtained from the pre- and post- (i.e. extant) European-colonisation biomass surfaces (Berry and Roderick, 2006) (termed B&R hereafter). Those layers were derived from growth modelling under past and present atmospheric CO_2 concentrations and from remote sensing; resolution was (0.05 x 0.05º, i.e. ~5.5 x 5.5 km pixels). Forest extents for those two periods were available in GIS polygon form at 1:1,000,000 scale (DEWR, 2007). The B&R layer of extant biomass represented net effects on biomass, such as from deforestation, land degradation, infrastructure, regrowth, improved pasture and woody-thickening. A higher resolution layer (0.0025 x 0.0025º, i.e. ~250 x 250 m pixels) of forest potential biomass was obtained from the Commonwealth Department of Climate Change and Energy Efficiency, as used in the National Carbon Accounting System (NCAS) and in the model FullCAM (Richards and Brack, 2004; Waterworth and Richards, 2008). The NCAS layer was of aboveground biomass. A value of 15% (representing root biomass) was added to the NCAS layer, this being an average of 10.5% for ash eucalypts (Feller, 1980) and 20% for temperate, high-biomass eucalypt forests (Mokany et al., 2006). This value was therefore conservative as (a) more- xeric environments (such as semiarid rangelands) have a higher root:shoot ratio, e.g. 1.84 and 0.642 for shrubland and savannah respectively (Mokany et al., 2006), and (b) the global average plant species root:shoot ratio is 0.333 (Niklas and Enquist, 2002), as used in Berry and Roderick

(2006). Effects of a higher root:shoot ratio derived from regional allometrics (Henry et al., 2002) was also tested.

One estimate of change in rangeland biomass due to commercial grazing was calculated by the B&R pre-commercial-biomass from the B&R extant-biomass layer. Another estimate was from multiplying the ratio of extant/pre-commercial from the B&R layers (i.e. relative change) by the NCAS layer and subtracting that from the NCAS layer, to get absolute change in biomass, resampled to 250 m.

9.3.2.2 Annual fluxes

Emission estimates were available for: (a) biomass loss with deforestation, savannah burning and livestock methane (Robertson, 2003); (b) diesel fuel usage (Rolfe, 2002); (c) biomass loss with deforestation (Henry et al., 2002); (d) biomass and soil pools modelling (Hill et al., 2006); (e) overgrazing of mulga lands (Moore et al., 2001); and (f) biomass loss with deforestation (DNRW, 2008). Error margins were not given for most of those estimates but typical variation between different reported values for each emission type was in the order of 30%— the error margin assigned in the present work when tallying emissions. National rangeland fluxes were tallied from contributions (a) to (f) above and from calculations of ΔSOC in the present work.

For the estimate of emissions from biomass due to QLD deforestation only the rangeland portion from DNRW (2008) was used— 10.2 Tg year^{-1} from 0.343 Mha year^{-1}. QLD emissions from deforestation are ~65% of the national total (Raison et al., 2009). Substantial rangeland woody-vegetation clearing in both NSW and NT in the form of thinning, fence line clearing, pasture development and mustering laneways was observed qualitatively. Clearing rates in NSW have recently increased (Fensham et al., 2011). Therefore if QLD deforestation rates decrease further, then the figure of 10.2 Tg year^{-1} will remain for several years equitable to a national total.

Estimates for C emissions from livestock excreta and savannah burning of 13 and 1.8 Tg $year^{-1}$ respectively (Cook et al., 2010) included undisclosed amounts from outside the commercial rangelands. Consequently, instead the rangeland-specific estimates of 8.2 and 3.5 Tg $year^{-1}$ (Robertson, 2003) were used.

SOC efflux following deforestation was estimated from earlier reports. Following deforestation in QLD the SOC to 1 m depth dropped by 3.42 Mg ha^{-1} over ~15 years (Harms et al., 2005). (That rate is not perpetual but would slow over two millennia.) Using the recent deforestation rate of 0.343 Mha $year^{-1}$ for QLD (DNRW, 2008), which was conservative when extrapolated over the past 30 years, gave a State-wide SOC emission from deforestation in QLD of 2.3(±0.7) Tg $year^{-1}$. Another estimate of annual SOC efflux from deforestation was 10% of annual lost biomass (Raison et al., 2009). That proportion is half of that calculated here but it may not have included previous deforestation and therefore would cease to be tallied if deforestation ceased; whereas the value proposed here includes effects from earlier deforestation and would continue for several centuries, although at a continually decreasing rate.

The fate of SOC in eroded soil worldwide constitutes an area of great uncertainty. With erosion in Australian rangelands there is a net loss of nutrients (Sparrow et al., 2003), which suggests that there would be a net SOC emission from eroded soil, rather than merely SOC translocation. Australian rangelands currently lose soil with a flux of 5.6 Mg ha^{-1} $year^{-1}$ (Loughran et al., 2004). Soil loss from the mostly un-commercialised sites is in the order of 1 Mg ha^{-1} $year^{-1}$, similar to the upper limit of soil formation (Loughran et al., 2004). Thus, a conservative estimate for soil loss, due to rangeland commercialization is 4.6 Mg ha^{-1} $year^{-1}$. Calculation of SOC efflux due to erosion Australia-wide was based on that earlier work and adjusted to fit a logistic curve (slower at onset and completion). Assuming a typical but conservative SOC concentration of 0.5 % by weight, and a conservative SOC loss-fraction from eroded soil of 20% (Lal, 2004; Lal et al., 2004) gives, for the 369 Mha of commercial rangelands, SOC efflux at 1.7(±0.9) Tg $year^{-1}$ (assuming a 50% error margin). Loss of dissolved organic carbon (DOC) from the soil was assumed to be part of the SOC erosion loss and was not modelled separately.

Two values for total annual emissions were calculated: (a) 'Total1' included erosion plus losses from recent deforestation and diesel usage, and (b) 'Total2' = 'Total1' + methane emissions from livestock + savannah burning.

To gauge the C fluxes associated with sequestration in the rangelands two scenarios were examined: (a) rehabilitation of degraded rangelands nationwide starting in 2011, using rate data from woody-thickening (Hibbard et al., 2003), and (b) regrowth of recently deforested land in QLD. For the purpose of showing the impact of ameliorative management, two subsets of rehabilitation were considered: (a) erosion ceased in 2050, and (b) erosion continued.

The process of sequestering SOC through establishment of woody vegetation in semiarid climates on degraded land is slow, with an asymptote near 2000 years (Hibbard et al., 2003). Error margins for SOC sequestration on degraded rangeland were high. Consequently estimates in the present work were intended as a guide to the order of magnitude.

9.3.3 Climate change influence on ΔSOC

The ΔSOC accompanying the changes in annual temperature and rainfall resulting from climate change was calculated, but without CO_2 fertilisation, following the same method as in Chapter 2. Comprehensive modelling of climate change found forest biomass to be most responsive to change in precipitation, and SOC more responsive to change in temperature (Cowling and Shin, 2006) but the link between biomass and SOC would mean that long-term change in rainfall would also affect SOC. Owing to the probably more-pronounced impact of climate change on forest ecosystems due to increased fire (Gonzalez et al., 2010) only the forested areas (using the DCCEE (2010) of forests, Chapter 1) were examined in the present work and not the non-wooded grasslands. Temperature and rainfall layers for the future climate were calculated using Ozclim (CSIRO, 2007) with settings from the global climate models 'ECHAM5/MPI-OM', 'Miroc-M', 'CSIRO Mk3.5' and 'ECHO-G';

under medium climate sensitivity and with the IPCC emission scenarios 'A1B' and 'A1FI'. The choice of climate models followed recommendations for Australia in Perkins *et al*. (2007).

SOC data to 0.3 m for Australia, for the year 2000, was available in GIS raster format (from Barson et al., 2002). The potentially forested rangelands contained 69% of the rangeland SOC to 0.3 m. SOC to 0.3 m depth for the years 2000 and 2100 was calculated using the formula from Bird et al. (2002) (Chapter 2, section 2.3). The calculated SOC layer for 2100 was divided by that for 2000 to give relative change. The relative change was multiplied by the SOC layer (Barson et al., 2002) to obtain the absolute value of SOC in 2100, to 0.3 m.

9.3.4 Regrowth in QLD

Remnant and non-remnant forest ecosystem data from vegetation mapping were available for 2003 from the QLD Herbarium (personal communication, Rosemary Niehus, 2009). Data were not available for some parts of western QLD, however deforestation had been mostly in the eastern, higher-rainfall country. The area of analysis was 22.66 Mha, consisting of 114,749 remnant ecosystem polygons. Extant biomass of regrowth was calculated from satellite-derived maps of foliage projected cover (FPC) of 25 x 25 m resolution, as of 2007 (personal communication; John Armston, Dave Harris, and Craig Shephard; QLD Dept. of Environmental and Resource Management; 2009). FPC was converted to stand basal area (formula from Armston et al., 2009) and stand basal area was converted to aboveground and below-ground biomass (formulae from Henry et al., 2002). Those three formulae had been developed for mature vegetation rather than for regrowth, which meant that calculated biomass values for juvenile regrowth (which can have early canopy closure) may have been too high. However, as the aim was to calculate missing biomass and hence sequestration potential, that aspect would provide an underestimate rather than overestimate, i.e. it would most likely provide a conservative estimate of possible C sequestration. Error margins were unavailable for

all equations used and uncertainty in the final results were calculated based on differences between different model outputs and between different initial datasets.

FPC and NCAS biomass values were compared for selected, remnant, regional ecosystems, because in those locations extant biomass should have equalled potential biomass. To reliably compare the NCAS and FPC biomass layers the following were excluded: areas deforested since 2002, mixed ecosystem types, and polygons smaller than 2 ha (to avoid edge effects). Also, a 100 m interior buffer for all ecosystem polygons was subtracted to accommodate the error margin of ±25 m in the FPC layer. Those exclusions left 506 remnant ecosystem polygons for comparison between biomass derived from FPC and NCAS. There was substantial variation for individual polygons but good agreement at the State-level. The average ratio of NCAS/FPC State-wide was 1.11 (SD 0.56, N=516). The difference from a ratio of 1 was not significant and therefore biomass derived from FPC and NCAS was treated as equivalent. The potential biomass of all regrowth forest polygons, State-wide, was subsequently obtained from the NCAS layer.

Biomass growth typically follows a logistic curve, but parameterization of such a curve requires knowledge of when growth began, which was not available for much of the regrowth. Alternatively, the growth until about 85% maturity can be roughly approximated as linear (e.g. Fensham and Guymer, 2009). Complete recovery of regrowth has been estimated to require 70 years (Fensham and Guymer, 2009), whereas 85% of maximum biomass for woody-thickening has been estimated at 140 years (Hibbard et al., 2003). As a compromise between those two studies it was assumed that 85% of mature biomass would be achieved in 100 years. Faster growth rates are likely where live root stock remains below ground after deforestation (e.g. lignotubers), but slower growth is likely from seed, especially on degraded land. The biomass present after 50 years was calculated assuming a linear growth rate. Where regrowth had been re-cleared, the most recent deforestation event was used as the starting point. An approximate, linear sequestration rate for QLD was calculated from the difference between the extant biomass and that in 50 years' time for all

regrowth polygons; those differences were summed and the total divided by 50 to get annual sequestration flux.

9.4 Results

9.4.1 Nationwide net ΔC to-date

Approximately 50% of Australia's potential forest biomass (from the NCAS layer) was in the commercial rangelands, approximately 50% of the rangelands were potentially forested (Table 9-2), and ~70% of the rangeland biomass (both extant and pre-commercial) was in the potentially forested rangelands (Figure 9-1, Table 9-1). Thus the potentially forested areas have the capacity to hold the majority of C in biomass and correspondingly, the majority of SOC (from the correlation presented in Chapter 7).

From the B&R layers resampled to 1000 m pixels the C in potential biomass was 6.50 Pg. The potential biomass from the NCAS layer was 6.59 Pg (Figure 9-1). Thus the two values for potential biomass differed by only ~2%.

The NCAS layer had approximately 20 times the spatial resolution of the B&R layer and arguably had more accurate pixel values. On close examination the NCAS layer had not differentiated much of the biomass of the Mitchell Grass downs (*Astrebla spp*.) of about 32 Mha in northern Australia from that of nearby forest. Biomass for those two vegetation types was however differentiated in the B&R layers. Thus when the ratio of extant/pre-commercial from the B&R layers was applied to the NCAS layer the calculated emission from the grasslands was higher than their pre-commercial stocks. Consequently, the NCAS layer was not further used in combination with the B&R layers. The high potential in the NCAS layer for the grasslands may have arisen due to the layer's independence of an environmental characteristic that gave preference for grass rather than tree cover (e.g. clay expansion/contraction or historical fire regime).

Table 9-1. Areas, carbon in potential biomass (from Richards and Brack, 2004) and deforestation (from DEWR, 2007) for Australian commercial rangelands, Australian forests and for Australia as a whole (last row).

Land-use/cover	Area (Mha)	C in Biomass (Pg)	% of rangelands		% of forests		% of Australia	
			Area	Carbon	Area	Carbon	Area	Carbon
Rangeland	369	6.593	100	100			49	33
Potential forest in rangeland	195	4.595	53	71	46	28	26	23
Potential forest in Australia	426	16.15			100	100	57	80
Deforested Australia	95.2	5.897			22	-37	13	-29
Deforested rangeland	21.9	1.050	11	-23	5	-7	3	-5
Australia	752	20.24					100	100

Table 9-2. Areas of potentially forested and never-forested Australian commercial rangeland and changes in their C in biomass since commercialization (derived from Berry and Roderick, 2006).

	Forest cover status	NSW	NT	QLD	SA	VIC	WA	Total	Total%
Area (Mha)	Never-forested	12.74	40.47	62.96	26.69	0.01	31.67	174.5	47
	Potentially-forested	18.75	29.77	74.03	16.28	0.20	55.42	194.5	53
	Total	31.49	70.24	136.99	42.97	0.22	87.09	369.0	100
C in biomass (Tg)	Never-forested	-13.58	-19.67	-17.40	-1.80	0.00	-16.43	-68.89	17
	Potentially-forested	-28.70	-50.61	-223.74	-3.90	-0.57	-37.25	-344.8	83
	Total	-42.28	-70.29	-241.13	-5.70	-0.58	-53.69	-413.7	100

The value used for C in potential biomass in the commercial rangelands was therefore taken from the B&R layers: 6.5(±0.5) Pg. That corresponds to approximately one quarter of Australia's total potential C in biomass (of 23.6 Pg), from approximately 50% of Australia's land area.

From the deforestation estimate using the national vegetation assessment (DEWR, 2007) and the NCAS layer, Australia had lost 1.05 Pg of C, being 23% of its forested rangeland biomass (Table 9-1).

From the B&R layers resampled to 1000 m pixels, the extant stock of C in biomass on the commercial rangelands was 6.08 Pg. Thus a net 0.418 Pg had been emitted. But from the B&R layers resampled to 250 m resolution the loss was only 0.414 Pg. That difference of ~1% was typical of error margins involved in GIS calculations using the national data sets. They were in part due to trimming of pixels when converting from large pixels (~5.5 km) in geographic coordinates (degrees) to projected coordinates (metres) and then overlapping with GIS layers originally derived in polygon form in projected coordinates. From close examination of the B&R layers the estimate of the errors in total extant biomass stock for the rangelands was ~15%, but up to ~50% for individual sites such as salt lake edges and coastal tidal zones. Those latter areas probably had high error margins due to problems of capturing variation in inter-annual or seasonal growth by remote-sensing.

The emission calculated from the B&R layers was a net value and therefore included all effects such as land degradation, deforestation, regrowth, woody-thickening and CO_2 fertilisation. Australia had lost 6% of its rangeland C in biomass due to commercial grazing, with 0.345 Pg (83%) of the loss from potentially forested rangeland and 0.069 Pg from never-forested rangeland (Table 9-2, Figure 9-2).

An average of the two methods of biomass loss— (a) NCAS-potential plus deforestation from DEWR (2007), and (b) difference between the B&R layers: (1.05 and 0.414 Pg respectively) — gave 0.73(±0.4) Pg for C efflux from biomass due to commercialization of the rangelands.

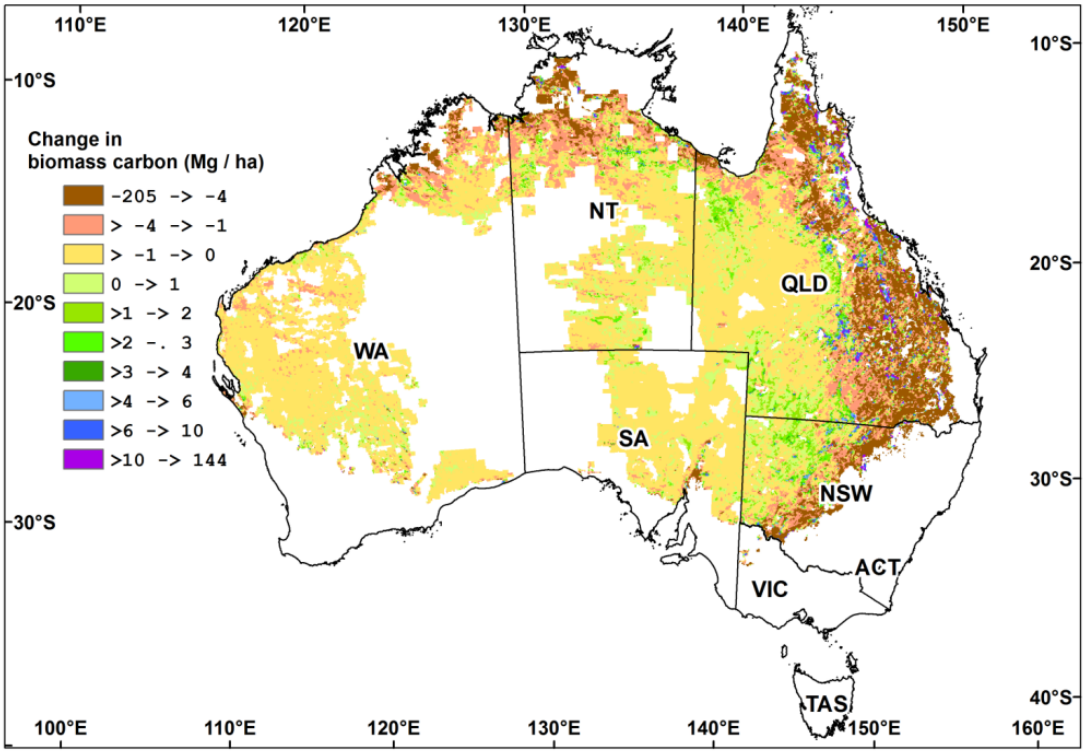


Figure 9-2. Change in C in biomass since onset of commercial grazing in the rangelands (derived from Berry and Roderick, 2006). Note that the major losses are from eastern QLD, and that deforestation is most often incomplete at the pixel level (0.05 x 0.05º, i.e.

The highest State or Territory biomass losses following commercialization of the rangelands were from QLD, from the mostly deforested eastern section (Table 9-2, Figure 9-2). QLD held more rangeland biomass than all the other States and Territories combined and 53% of that QLD biomass was on freehold land (in 27% of the area of QLD rangelands), which contained more biomass than any of the other States individually.

From the modelling of soil erosion the estimate of SOC rangeland losses to-date was 0.086(±0.022) Pg and from deforestation to-date the SOC loss was 0.07(±0.04) Pg—a total SOC efflux of 0.16(±0.05) Pg. Qualitatively it was found that erosion appeared more pronounced in overgrazed areas and near tracks, (both vehicle and livestock tracks), and especially pronounced near where tracks crossed riparian zones, extending upstream from the crossing. As a qualitative observation, the

consequent erosion in riparian zones caused the highest, localised negative impact on trees and on soil stability (Figure 9-3).

Of 602 randomly selected watering points across ~6 Mha in the southern Northern Territory, 31% were within 100 m of a creek and 68% were within 500 m of a creek (Figure 9-4). Thus there appeared to have been a distinct preference for managers to locate livestock watering points near creeks. Also, natural watering holes are inherently centred in riparian zones. The majority of herbaceous feed in dryer times is in riparian zones, which naturally attracts higher concentrations of livestock. Qualitatively, that spatial concentration of impact was more prominent during dry periods prior to rain, and especially during droughts. Throughout most of semiarid Australia, where woody biomass was sparse, the biomass was often higher in riparian zones, due to topographic flow (e.g. Figures 9-5, 9-6). Both the location of watering points and the concentration of available feed contributed to high riparian-zone impact.

In several experimental [livestock] exclosures established in the 1960's in WA I observed qualitatively that buffel grass (*Cenchrus ciliaris* L. (syn. *Pennisetum ciliare* (L.) Link)) had much higher biomasses than outside the exclosures, especially under shrubs and trees. Buffel grass is an African species introduced into Australia for livestock forage and to reduce soil instability resulting from overgrazing, following similar introduction into North America. Buffel biomass was also high in riparian areas (Fig. 7) although it was reduced to some degree by livestock grazing. However, in the absence of high livestock numbers, and where native macropod (kangaroos and wallabies) numbers were high, the buffel biomass appeared sufficiently high to represent an increased fire risk. Near where grazing properties bordered rural townships that risk was sometimes reduced by prescribed burning, but that activity also reduced biomass of some native mature trees.

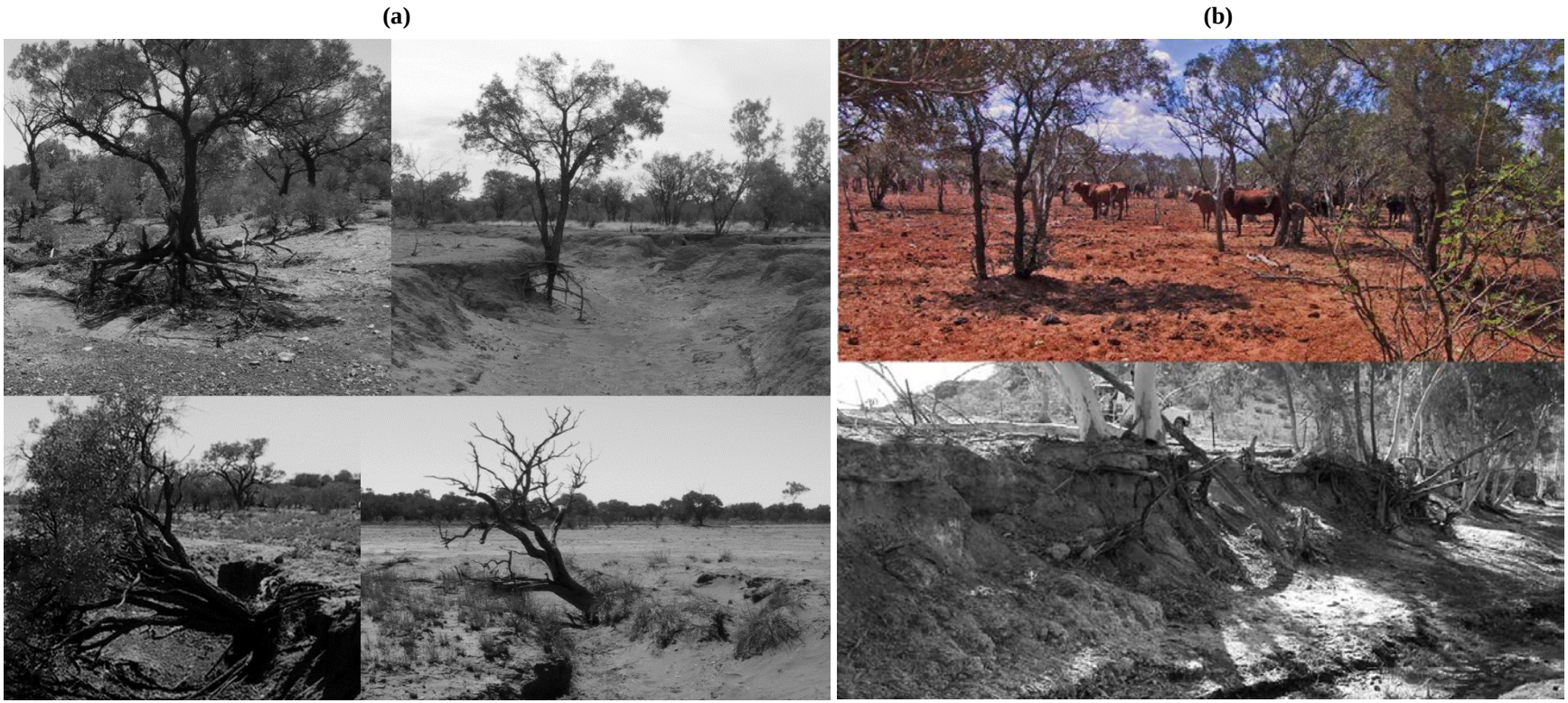


Figure 9-3. Examples of slow riparian biomass- and SOC-loss from overgrazing and tracks, southern Northern Territory. **(a)** Erosion and stages of gradual biomass decline due to over grazing, combined with a vehicular crossing downstream. Note persistence of buffel grass in lower right photo. **(b)** Degradation from cattle at natural waterholes. Upper photo: widespread soil disturbance, no litter cover, no young Gidyea (*Acacia cambagei*) but colonisation by a less-palatable thorny acacia species (lower right). Lower photo: bank erosion and tree decline around a spring, including degradation from both cattle and feral herbivores.

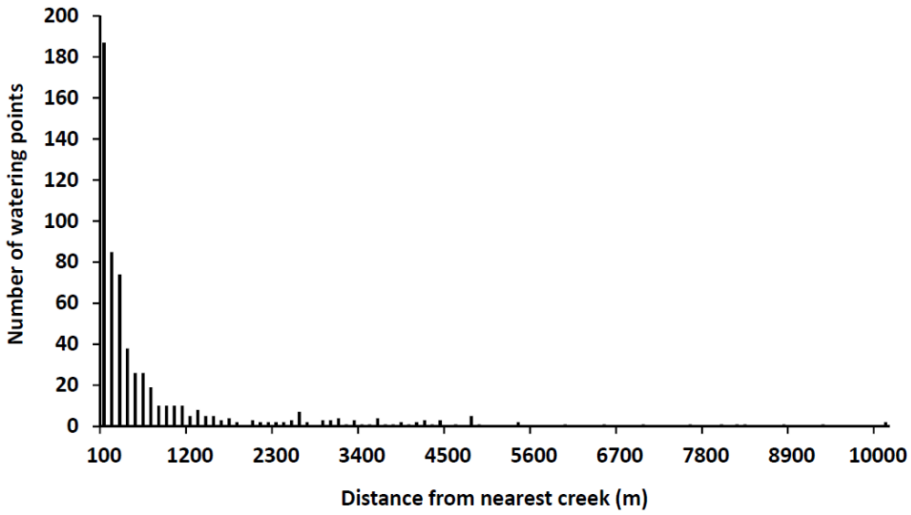


Figure 9-4. Histogram of number of watering points vs. distance from nearest creek, for ~6 Mha of the Northern Territory. Note the concentration close to creeks.

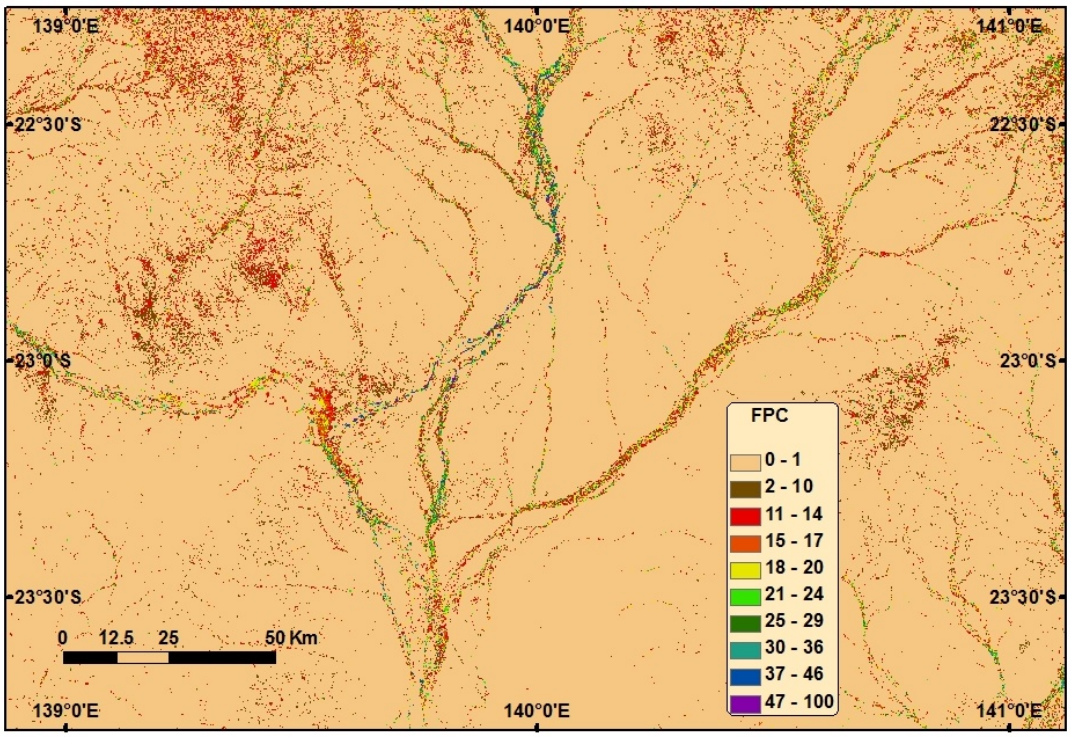


Figure 9-5. Example distribution of forest biomass shown as FPC in minimally forested rangelands. Biomass is concentrated in riparian zones. FPC values > 10 are forest. This region is west of the most concentrated deforestation. The upper half of the map is the Mitchell Grass downs ecosystem type and the lower half is the Channel Country ecosystem type— similar ecosystems are widespread.

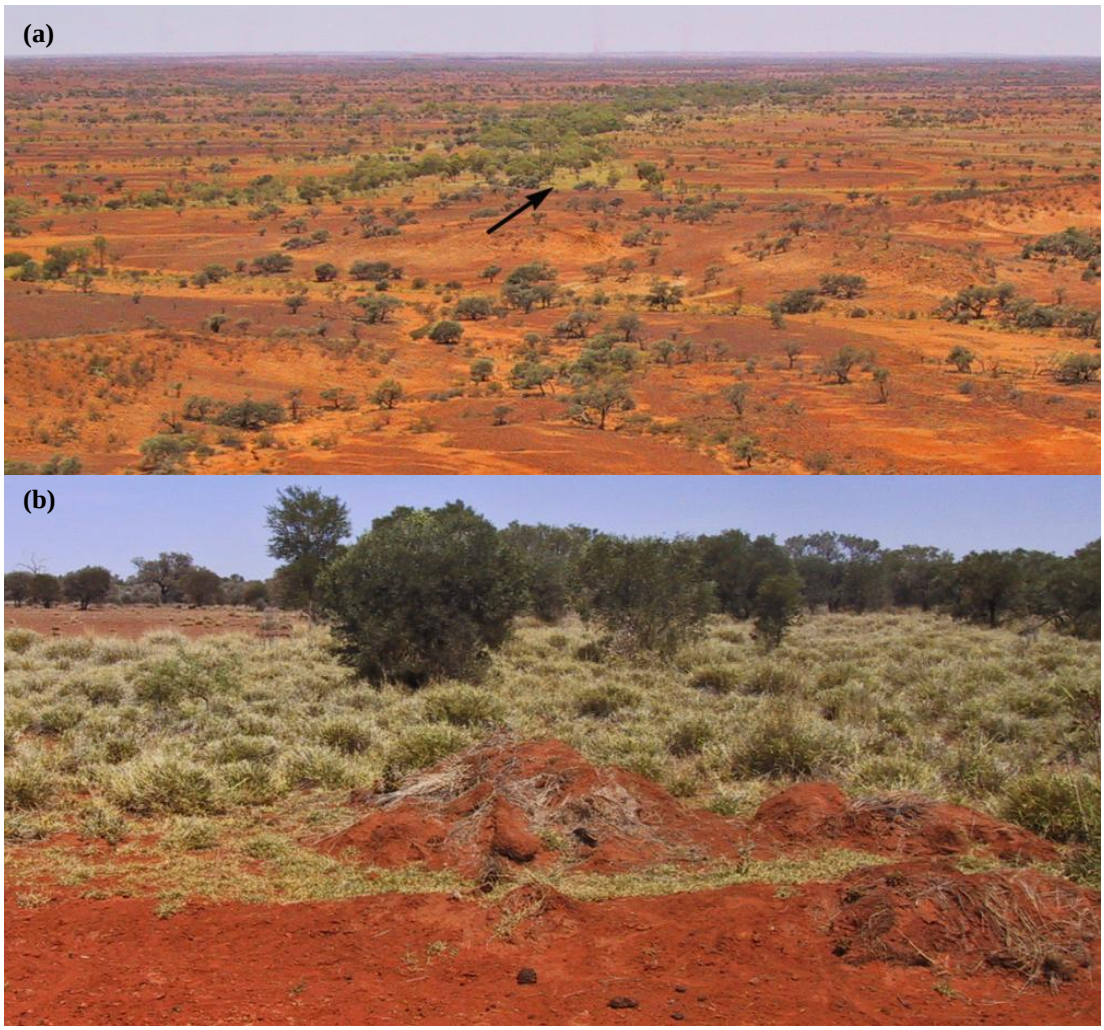


Figure 9-6. Distribution of forest biomass relative to riparian zones in minimally forested rangelands. **(a)** biomass distribution typical of ecosystems similar to the 'Channel Country'. The lighter-green trees are in moister riparian zones (top centre), the pale green shade on the soil surface in the riparian zone (e.g.: arrow) is buffel grass, medium green are the eucalypts, and the darker grey-green are acacias. **(b)** typical riparian buffel infestation in commercial grazing land of 'Channel Country'. Considerable high dry biomass level, during drought period, but with green tips due to recent light showers.

Table 9-3.a Extant (2000) SOC (to 0.3 m), stocks in 2100 and percentage change for the forested rangelands from different climate models and emission scenarios.

State	In 2000		CSIRO-Mk3.5				ECHAM5				ECHO-G				Miroc-M			
			A1B		A1FI		A1B		A1FI		A1B		A1FI		A1B		A1FI	
	Tg	%	Tg	Δ(%)	Tg	Δ(%)	Tg	Δ(%)	Tg	Δ(%)	Tg	Δ(%)	Tg	Δ(%)	Tg	Δ(%)	Tg	Δ(%)
NSW	222	8	107	-52	63	-72	145	-35	104	-53	229	3	226	2	217	-2	211	-5
NT	417	14	192	-54	105	-75	270	-35	197	-53	336	-20	291	-30	356	-15	320	-23
QLD	1523	51	818	-46	544	-64	1005	-34	775	-49	1342	-12	1273	-16	1325	-13	1244	-18
SA	182	6	38	-79	15	-92	99	-46	65	-64	151	-17	133	-27	168	-8	159	-13
VIC	6.0	0.2	3.1	-49	1.5	-75	4.9	-18	4.3	-29	5.6	-5.8	5.4	-10	5.7	-4.4	5.5	-7.3
WA	610	21	174	-71	88	-86	343	-44	226	-63	343	-44	216	-65	402	-34	297	-51
Total	2961	100	1332	-55	817	-72	1868	-37	1371	-54	2406	-19	2145	-28	2474	-16	2236	-24

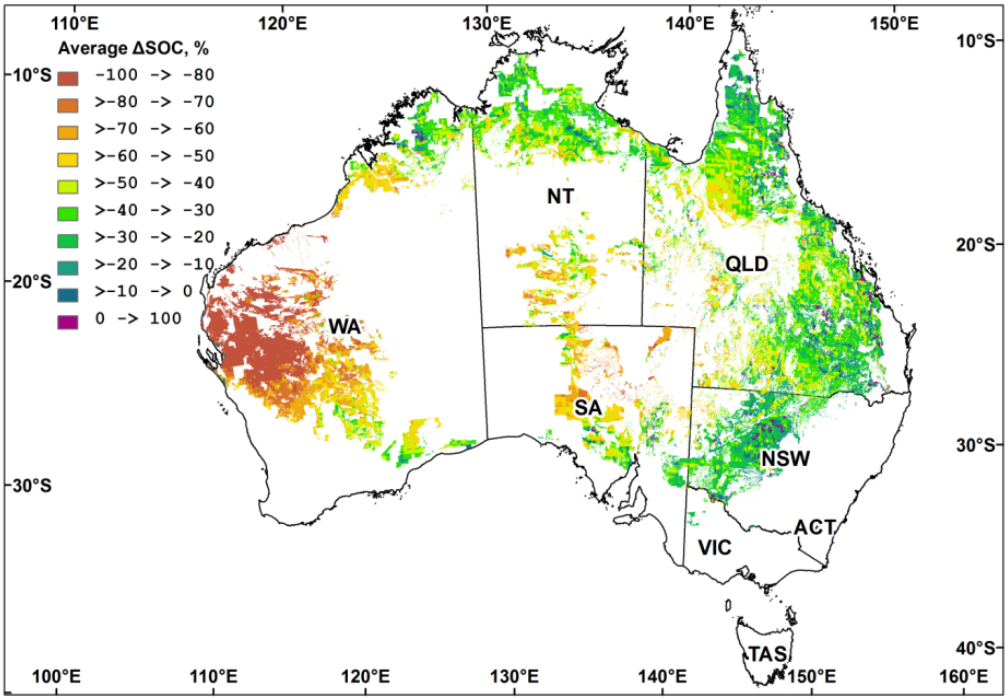
Table 9-3.b State-wide totals for SOC for 2100 (to 0.3 m) in forested rangelands. SD (in brackets) assigned from variation across four models and two emission scenarios.

State	A1B		A1FI		Average	
	Tg	Δ(%)	Tg	Δ(%)	Tg	Δ(%)
NSW	175	-21	151	-32	163(66)	-27(11)
NT	289	-31	228	-45	258(86)	-38(13)
QLD	1123	-26	959	-37	1041(301)	-32(9)
SA	114	-37	93	-49	104(99)	-43(24)
VIC	4.8	-19	4.2	-30	4.5(1.5)	-25(8)
WA	318	-48	207	-66	261(104)	-57(23)
Total	2020	-32	1642	-45	1831(597)	-38(12)

9.4.2 Climate change influence on ΔSOC

The forecast ΔSOC with climate change for the forested rangelands (Figure 9-7, Table 9-3) showed substantial variability owing to (a) differences between climate models, (b) differences between global emission scenarios, and (c) areas with low extant stocks accruing high variability between 2000 and 2100 from the equation of Bird *et al*. (2002). The largest differences nationally were between the models CSIRO-Mk3.5 and Miroc-M, mainly due to differences in forecast rainfall in northeast Australia. The area of most consistency between models was the central-west of WA (known as the Pilbara and Gascoyne), where all models forecasted a drop in rainfall. Owing to the consistency for that area its average forecast when all four models were combined was large, as there was no superposition of differently signed fluxes. In areas where different models diverged most, the net average ΔSOC was closer to zero. SA showed the highest standard deviation, due to the lower, average, extant SOC inducing more relative variability.

(a)



(b)

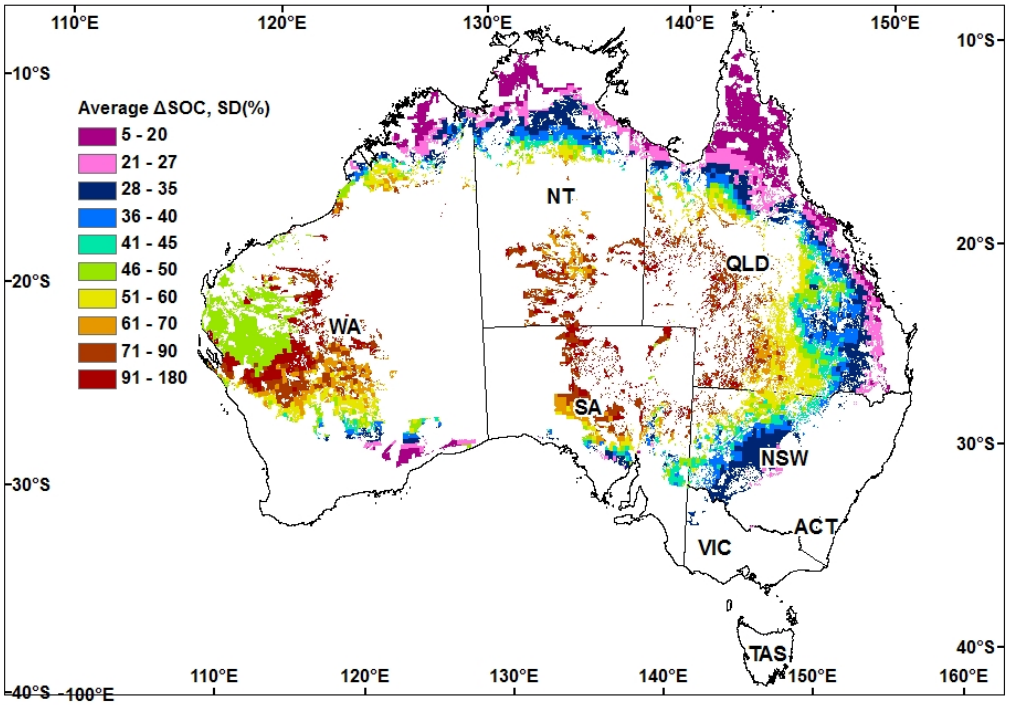


Figure 9-7. Change in SOC to 0.3 m with climate change, for the forested commercial rangelands between 2000 and 2100. Average from the four climate change models and two emission scenarios. **(a)** Percentage change. **(b)** Standard deviation (as a percentage) of the average.

9.4.3 Annual emissions

The annual emissions calculated were compared with those from the literature (Figure 9-8). Nationally, the two-highest singular emissions were biomass loss from deforestation in QLD and nationwide methane emissions from livestock (the latter is the main difference between 'Total2' and 'Total1'). Even though ΔSOC was a relatively minor flux on an annual basis (comprising -2.3 Tg year^{-1} from deforestation and -2.1 Tg year^{-1} from erosion) it was half the efflux from biomass from deforestation in QLD and equivalent to 3% of Australia's Kyoto-Protocol annual GHG emissions (totalling 157.32 Tg year^{-1} of C-equivalent (DCCEE, 2008)). Furthermore, ΔSOC takes longer than biomass emissions accompanying deforestation (due to the multi-centennial half-life of some SOC pools e.g. Liao et al.

(2006a)) and thereby ΔSOC will accrue a higher total emission in the long-term, than the short-term -2.3 Tg year^{-1}. Also, if national destocking was instantaneous the land degradation would not cease immediately and SOC efflux would continue. Thus, the SOC efflux is one of the major effects.

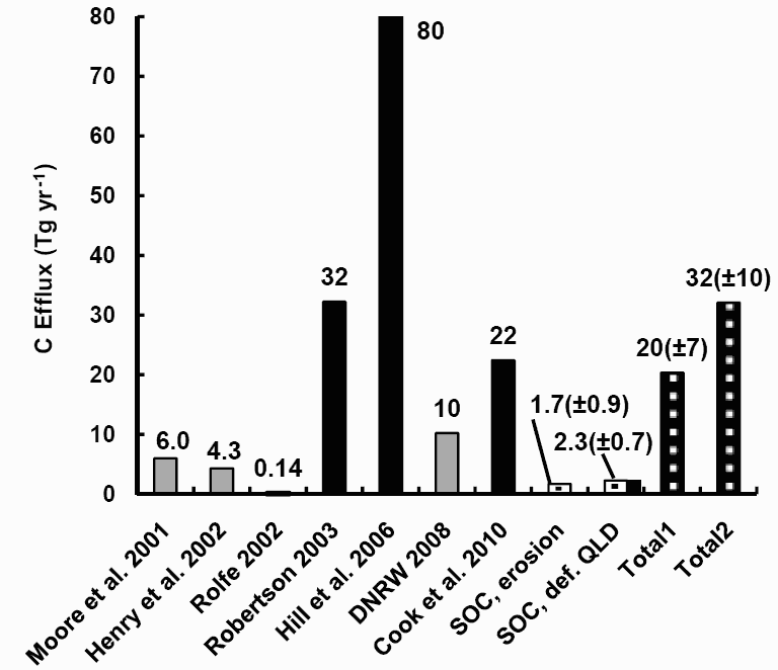


Figure 9-8. Estimates of annual emissions from Australian rangelands. Lighter columns represent only some component pools, and darker columns represent all pools. Values from the present work are the right hand four columns.

From the approximation for nationwide rangeland soil rehabilitation (Figure 9-9) if erosion was allowed to continue while rehabilitation that started in 2011 was ongoing, then the total SOC emission (to 0.3 m depth) due to commercialization of the rangelands would be in the order of 1.3(±0.5) Pg, peaking in 2400, including 1.2(±0.5) Pg after 2011. The error margins for those SOC flux estimates are high (possibly around 90%) because the calculation was based on single values for the national SOC loss by erosion, the percentage of C in rangeland soil, and the loss rate of SOC from eroded soil.

If rehabilitation began in 2011 and all erosion was curtailed by 2050, then the SOC efflux since onset of rangeland commercialization would be limited to 0.15(±0.14) Pg, and 0.062(±0.056) Pg from 2011–2050. Thus, a SOC emission of 1.2-0.062= 1.16(±0.5) Pg could be avoided by initiating rehabilitation in 2011 and ceasing anthropogenic erosion by 2050.

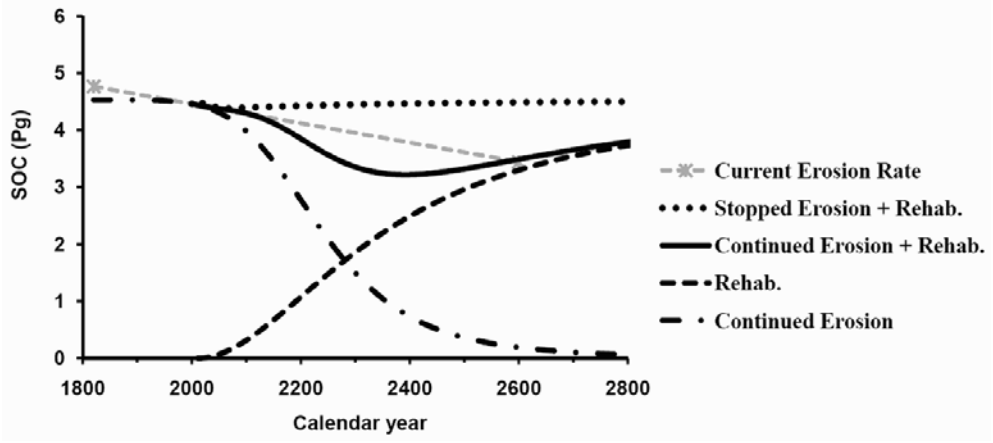


Figure 9-9. SOC for all Australian rangeland (to 0.3 m) with current erosion rate and future rehabilitation. Emissions are much less if erosion is arrested in 2050 than if erosion is allowed to continue concomitant with rehabilitation. Recovery time is in the order of 2,000 years owing to the inherently slow rate of SOC sequestration.

9.4.4 Sequestration in QLD regrowth

The area of regrowth forest (Figure 9-10,11, Table 9-4) in the commercial rangelands of QLD was 22.7 Mha., i.e. 29% by area of QLD's potentially forested rangelands had potential to sequester C via forest regrowth; which corresponds to 16% of all QLD's rangelands. The largest regrowth cohort (by area) was that with only 10–20% regrowth of potential biomass present (Table 4a). Deforestation in QLD to date corresponded to ~92% of rangeland deforestation nationwide and was therefore representative of rangeland deforestation nationally.

Figure 9-10. Forest change in southeast QLD's commercial rangelands. **(a)** fresh broad-scale clearing, fraction of potential present is zero. **(b)** young regrowth forest amongst multi-event, patchy clearing.

The highest potential was consistently in the higher rainfall zone in eastern QLD (Figure 9-11). The spatial distribution of potential sequestration was more dependent on the clearing activity than on the rainfall gradient, because eastern QLD generally had a lower fraction of its potential present. That spatial distribution may have been due to more re-clearing events in the east, accrued over a longer period of usage or more intensive usage in the east. Average property size was smaller in the higher rainfall areas generally across Australia, which may have been an indicator of the intensity of land-use.

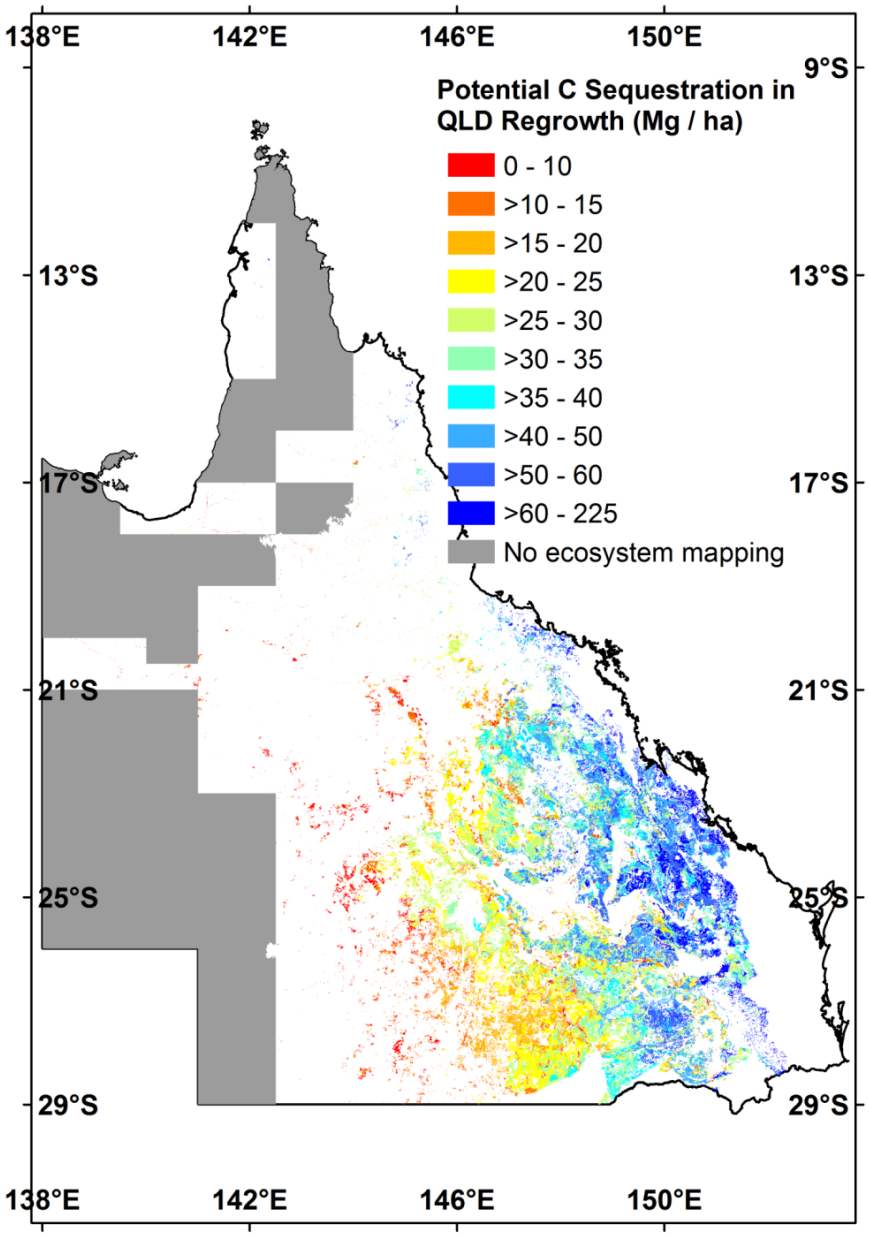


Figure 9-11. Map of potential C sequestration in QLD regrowth forest. Geographic projection.

From the employed approximations the errors for QLD rangeland regrowth were ~30% for extant stocks and ~50% for sequestration rates. The extant C in biomass of the regrowth was 0.179(±0.054) Pg and it could reach a maximum of 0.974(±0.292) Pg: the average regrowth forest was at ~30% of its maximum biomass. The C efflux

from biomass from QLD deforestation to date was therefore 0.794(±0.397) Pg (Table 9-5), which was the amount that theoretically could be recovered as sequestration.

The C sequestration rate (with the linear growth rate approximation) was 0.36(±18) Mg ha^{-1} $year^{-1}$. If all the extant regrowth forest was allowed to mature then the C sequestered in biomass over 50 years would be 0.410(±0.205) Pg, i.e. a sequestration rate of 8.2 (±4.1) Tg $year^{-1}$. That rate assumes no detriment to growth from climate change effects, such as increased water deficit or fire risk.

Table 9-4. Carbon sequestration capacity in biomass in regrowth forests on commercial rangeland in QLD, partitioned into areas in terms of percentage of potential present.

Extant biomass (% of potential)	Area (Mha)	Emitted carbon= potential– extant (Tg)
0–<10	2.1526	127.28
10–<20	9.3213	398.30
20–<30	4.5299	139.55
30–<40	3.3501	73.96
40–<50	2.1486	40.04
50–<60	0.5610	10.13
60–<70	0.2187	2.87
70–<80	0.1341	1.41
80–<90	0.1046	0.70
90–100	0.0484	0.37
Total	22.57	794.6

Table 9-5. Carbon sequestration capacity in biomass in regrowth forests on commercial rangeland in QLD, and growth rate. Emitted C is what can be sequestered.

Extant	0.179(±0.054) Pg
Potential	0.974(±0.292) Pg
Emitted= potential- extant	0.794(±0.397) Pg
Sequestration over next 50 years	0.410(±0.205) Pg
Sequestration rate (next 50 years)	8.2(±4.1) Tg $year^{-1}$
Sequestration flux (next 50 years)	0.36(±0.18) Mg ha^{-1} $year^{-1}$

The regrowth calculations above assumed a root:shoot ratio of 0.15. The regional allometrics from Henry *et al*. (2002) for aboveground and below-ground biomass, when applied to the remnant ecosystems, yielded an average root:shoot ratio of 0.33. Using that higher ratio gave a C efflux from past QLD deforestation of 0.939(±0.469) Pg, i.e. 18% higher than 0.794(±0.397) Pg. However if 0.33 was used as the root:shoot ratio then the assumption of NCAS biomass being equal to FPC biomass was no longer valid, because the NCAS/FPC biomass ratio of 1.11 became 1.28, indicating that NCAS biomass values may be too high. The root:shoot ratio was thus a major cause of uncertainty and it was reasonable to retain the initial estimate of the C that could be sequestered in biomass, by maturation of regrowth, at 0.794(±0.397) Pg.

9.5 Discussion

9.5.1 Emissions

Under commercial grazing and climate change the forested rangelands provide major rangeland C fluxes. The effect on C stocks from commercial grazing in the forested rangelands is similar to the global trend for 'woodland' type rangeland (Ellis, 2011). Four main avenues to reduce C emissions from rangelands have been recommended (Henry et al., 2002): (a) reduced deforestation, (b) protection and enhancement of SOC, (c) reforestation, and (d) increasing the scientific knowledge involved in these processes. For the first three avenues it was in the present work found that: (a) an emission of 10(±3) Tg year^{-1} could be avoided, (b) an emission of 1.53(±0.5) Pg of SOC could be avoided by rangeland rehabilitation and cessation of erosion, and (c) 0.794(±0.397) Pg of C could be sequestered in biomass in QLD regrowth (or 8.2(±4.1) Tg year^{-1} of C over the next 50 years). The fourth avenue is discussed in more detail below.

The discrepancy between the C in biomass emission estimate of 0.414 Pg from all commercial rangeland using the B&R layers (Table 9-2) and the 1.05 Pg from

deforestation alone using the NCAS layer and DEWR (2007) (Table 9-1) was most likely due to a combination of factors: (a) deforestation being incomplete within any one 0.05x0.05º pixel; (b) NDVI (normalised vegetation index) of young regrowth or its grass replacement being similar to or greater than the NDVI of mature forest; and (c) possible overestimation of CO_2 fertilisation in Berry and Roderick (2006). The B&R layers inherently included biomass loss due to, for example, overgrazing, fence-line clearing, laneway clearing, firewood collection, cutting for fodder and small-scale illegitimate felling; and biomass gain due to thickening (although this has been shown to be minimal on net for Australia (Witt et al., 2006; Fensham et al., 2011)); whereas the method of determining biomass change from the DEWR (2007) and NCAS layers referred more to broad-scale deforestation alone. Additionally, if a root:shoot ratio of 0.333 had been used in place of 0.15 for the deforestation estimate the emission would be 1.22 Pg rather than 1.05 Pg. The magnitude of the loss of C in biomass (0.73(±0.40) Pg) due to rangeland commercialization can be compared with the nationwide loss of 3.3 Pg of C (from Berry and Roderick 2006) of which the major component was deforestation of arable land (with higher average biomass) for intensive farming, forestry and infrastructure.

Higher precision data, in the form of regional allometrics and 25 m pixels, yielded a C emission from biomass due to deforestation in QLD of 0.79(±0.40) Pg (Table 9-5), which was higher than the nationwide rangeland emission [from all sources] of 0.73(±0.4) Pg calculated from lower precision data. Furthermore, from the higher precision calculations, the average sequestration potential for QLD regrowth forest was 35 Mg ha^{-1}, but from the B&R layers that value was 3.4 Mg ha^{-1}; i.e. nearly an order of magnitude lower. Those comparisons suggest that the value for national rangeland biomass loss using the B&R layers was too low, and therefore that the calculated value of 1.05 Pg of C emitted to date from deforestation (Table 9-1) was more accurate. As neither could be verified further with the available data, there was no reason to nominate an alternative value for C emission from biomass due to deforestation in QLD, other than the 0.73(±0.40) Pg.

The net C efflux that accompanies erosion, due to commercialization of the rangelands, is dependent on: (a) the net balance between rates of loss and soil formation (pedogenesis), (b) the contribution of biomass growth to soil carbon formation with and without commercialization, and (c) erosion rates with and without commercialization.

Determining the fraction of rangeland erosion that is natural and that which is anthropogenic is not straightforward (e.g. Dregne, 1995). For the erosion estimate in the absence of commercialization I used the soil loss rate from minimally disturbed sites (1 Mg ha^{-1} $year^{-1}$), which is similar to the maximum rate of soil formation (pedogenesis) (Loughran et al., 2004), i.e. pedogenesis may just keep up with natural erosion. That gives the portion of rangeland erosion (of the 5.6 Mg ha^{-1} $year^{-1}$ nationwide) that is anthropogenic as 80%.

Evidence for soil loss due to commercialization is from: (a) historical, qualitative reports; (b) current fence-line contrasts in paddock soil condition; (c) point source erosion from anthropogenic activity; and (d) quantification of broad-scale anthropogenic contributions to wind erosion.

Degradation of Australia's rangelands due to grazing practices has occurred since the onset of commercialization in the 1850s, when overstocking was widely practised (Wheeler and Hutchinson, 1973; Harrington et al., 1979; Williams and Calaby, 1985; Barnes, 1997; McKeon et al., 2004). After a decade of commercialization there was substantial erosion, loss of perennial grasses and saltbush, shrub removal and salinity; and within 30 years there was woody shrub proliferation and scalded plains (Barnes, 1997; Scott, 2001). Severe erosion continued for several decades. During droughts, denudation and erosion prevailed near watering points, and vegetation further declined with topsoil lost in the wind. Stocking rates were maximised until negative effects were detected, and reduced when short-term economics became unsustainable (Barnes, 1997; James et al., 1999). Degradation was initially focussed around natural watering points (Little, 1964; Beard, 1990) but became more diffuse with bore usage, financial setbacks of overgrazing (with regrowth on defunct

properties), vehicular tracks, shrub cutting for fodder, and deforestation. Improvements in land condition were evident from the mid twentieth century, after the introduction of stocking limits (e.g. Barnes, 1997; Watson et al., 2007). Today amelioration of erosion is reliant on the reporting of the most prevalent degradation effects, which are qualitatively assessed (on-ground and by remote sensing), mostly from long-term changes in perennial grass cover or in shrub demographics (e.g. Dean, 2005; Watson et al., 2007). Soil carbon or biomass carbon stocks *per se* are unmonitored, other than change in biomass carbon in QLD from broad-scale deforestation. Change in soil carbon due to deforestation is longer-term and currently unmonitored during routine pastoral lease inspections, nationwide.

Contemporary assessment suggests that vegetation degradation and erosion affects over 55% of Australian commercial rangeland, with portions for the different States being 100%, 80%, 50%, 40% and 40% for NSW, SA, QLD, WA and NT respectively (Woods, 1984), including for example on 85.3 Mha within western QLD (Dawson and Turner, 1982). The rate of soil loss is estimated to be currently greater than soil formation in 70% of Australia's rangelands (Loughran et al., 2004), and >1 Mg ha^{-1} $year^{-1}$ (the upper limit of pedogenesis) in over 84% of WA, with highest rates in regions where pedogenesis is low (George, 2001). The most quantitatively rigorous pre/post-commercialization comparison for Australia is the "accelerated erosion index", which showed that wind erosion is higher with commercial rangeland grazing for over the half of QLD that is naturally susceptible to dust storms (Leys et al., 2001).

Ground cover (including plant and tree litter, grass, and microbiotic crusts) stabilizes soil against water and wind erosion, and can be used as a surrogate of soil condition, with SOC efflux potentially doubled in its absence (Perry, 1977; Eldridge and Greene, 1994; Shao et al., 1994; Roose and Barthès, 2006; Zuazo and Pleguezuelo, 2008). Soil erosion increases with decreased cover levels, such as through overgrazing, and cover levels >10–40% are necessary to prevent increased soil movement (Miles and McTainsh, 1994; McIvor et al., 1995).

Annual water runoff is at least doubled with conversion to exotic pasture after deforestation (with generally higher runoff in heavy pluvial events), and consequently erosion is also increased (Scanlan et al., 1991; Thornton et al., 2007; Zuazo and Pleguezuelo, 2008). The frequency and intensity of heavy pluvial events is forecast to increase with climate change— a trend evident in central Australia since *c*. 1970 (Pickup, 1998; Gallant et al., 2007; Pittock, 2009). The number of deforested rangeland sites measured in the continental study by Loughran *et al.* (2004), under-represented the deforestation in QLD, NSW and NT and therefore that estimate of erosion is possibly low for Australian rangelands as a whole. That would make the estimate of SOC loss calculated here with rangeland erosion most likely conservative.

Eroded soil is either carried by wind or water, with these vectors prevailing to different degrees. In each case there are three locations relevant to quantifying SOC flux: the erosion site, the transit route and the depositional site. Although the fate of SOC eroded by commercial rangeland grazing has been widely studied (e.g. Butler et al., 2003; Lal, 2004; Paull et al., 2006; Polyakov and Lal, 2008) it remains a topic of current scientific research, with a major concern being whether it represents a net loss to the rangeland system, or a redistribution.

Erosion from overgrazing, deforestation and various methods of attrition of woody vegetation incur a range of changes in soil properties which negatively affect biomass production (examples in: Christian and Perry, 1969; Harrington et al., 1979; Dawson and Turner, 1982; Murrell, 1984; Pickup and Chewings, 1986; Pickup, 1990; Chartres et al., 1992; McMahon et al., 1992; Sparrow et al., 1997; Kinloch and Friedel, 2005). Conversely, impacts severe enough to limit microbial community dynamics can limit soil respiration (limiting SOC efflux) (Grandy and Robertson, 2007), although under these circumstances, any emitted SOC is not replenished, due to an absence of biomass.

Biomass and nutrients are lowered in eroded areas (unless woody plants can re-colonise) and biomass can be higher in depositional areas where nutrients

accumulate. However the nutrients and water in the deposition zone can become decoupled from vegetation, with net nutrient loss, and the biomass on the relatively small deposition zone does not balance that lost from the eroded area (Sparrow et al., 1997; Sparrow et al., 2003; Tongway et al., 2003). A net nutrient loss involves C efflux (for organic nutrients).

The fate of the transported carbon will depend on oxygen availability (such as through physical agitation, moisture and oxidants) and temperature. Eroded SOC in sediments in aquatic systems contributes to the net carbon stock (Jacinthe et al., 2001; McCarty and Ritchie, 2002; Smith et al., 2005; Paull et al., 2006; Smittenberg et al., 2006). Emissions therein increase with increasing temperature, (with rates being substrate dependent) and within microbial tolerance limits (Weston and Joye, 2005; Michael Doering et al., 2011). Anoxic, waterlogged soils and lower temperatures, decrease the rate of organic decomposition (Trumbore, 1997; McCarty and Ritchie, 2002; Stolbovoi, 2002; McIntyre et al., 2009). Salt lakes and wetlands Australia-wide, on commercial rangeland properties and conservation reserves downstream of commercial rangelands, are examples of such anoxic sites. Other catchments drain seaward through major rivers.

There is preferential erosion of the surface soil and organic fractions of soils (which are lighter), and of microbial carbon, and thus the translocated carbon may be more labile than that which is not eroded (Gregorich et al., 1998; Mora et al., 2007). In that situation, if the deposited SOC has no additional protection mechanism, it can have two to four times the efflux than of the eroded site (Mora et al., 2007). Translocated SOC can incur fast initial losses, e.g. 15% within 0.27 years, at least ~12% emitted in transit and with higher losses for longer transit times, and less than half the translocated SOC may reach riparian transport (Polyakov and Lal, 2008). Carbon losses per unit mass are more from the larger particles than from finer fractions due to scission of the larger aggregates, preferentially where the carbon is located and the scission is achieved by flow-related shear stresses and raindrop impact, providing surfaces for microbial attack (*ibid*).

Jacinthe et al. (2002) found SOC losses of 29–35% from eroded soil, with nearly half of that lost within 0.055 year. Lal (2001) and Jacinthe et al. (2001) estimated that 20% of translocated SOC is emitted as CO_2. For net SOC efflux from soil loss, that value of 20% (which was used) would thus appear to be conservative (Lal et al., 2004) and therefore appropriately used in this study.

The calculation in the present work of a SOC efflux of 1.7(±0.9) Tg year^{-1} owing to net soil loss from erosion for the commercial rangelands, based on the soil erosion work of Loughran et al. (2004), does not include SOC emitted with erosion where the translocation is only within single study sites (which ranged from 400–3500 m). I.e. where there was soil movement but no net soil loss— those emissions are additional. This constitutes another reason for the SOC efflux with erosion calculated here to be considered a conservative value.

Similar calculations covering similar expanses and climates elsewhere are scarce. However, for Rajasthan (India) in the 200-500 mm rainfall zone Singh et al. (2007) found continually declining SOC due principally to land management effects. From 1972–2002 their SOC efflux averaged 0.49% year^{-1} to 1 m in the arid region, ranging from 0.14–1.1% year^{-1}, dependent on soil type. Assuming an intermediate emission of 0.6% year^{-1}, and applying that to the most degraded of Australia's rangeland (i.e. 5% of the 369 Mha) then the annual C efflux from the top 0.3 m of Australian rangeland is 11 Tg year^{-1}. That value is about six times the 1.7 Tg year^{-1}, calculated here but within expected error margins considering the approximations used. The range of values provides another indication of likely error margins.

The calculated annual C emissions ('Total1' and 'Total2') from the commercial rangelands of 20(±7) Tg year^{-1} and 32(±10) Tg year^{-1} are equivalent to 13(±4) % and 21(±6)% of Australia's Kyoto-Protocol annual GHG emissions, i.e. approximately identical to the reported Agriculture+LULUCF sectors' emission (DCCEE, 2008). 'Total2' is identical to that of Robertson (2003), although he included a high component from deforestation (20.5 Tg year^{-1} from the 1999 national GHG inventory), which was replaced in the tally in the present work by the efflux from

SOC and grazing of the mulga lands (*Acacia anuera* shrubland often around 3 m high, occupying 170 Mha) (not included in his estimate).

Australian national GHG emissions increase annually, and the present work the 2008 value of 553 Tg of CO_2-e (DCC 2009a) was used. They are currently reported according to Kyoto Protocol rules, however for the land sector this only includes agricultural emissions and net emissions from reforestation, afforestation and deforestation. Additionally ΔSOC is only included when the party concerned chooses to include Tier-3 carbon fluxes. Australia has elected not to include Kyoto Article 3.4 land use activities in its Kyoto account. Thus, several of the emissions calculated here are not currently reported and hence several major fluxes are omitted.

9.5.2 Sequestration

9.5.2.1 Sequestration in biomass

Moore *et al*. (2001) proposed that rehabilitation (by destocking and fire suppression) of the mulga lands could sequester 0.273 Mg ha^{-1} $year^{-1}$ in biomass and necromass carbon over ~60 years. Garnaut (2008) proposed C sequestration at 68 Tg $year^{-1}$ over "several decades" for "restoration" of the mulga lands, corresponding to 0.4 Mg ha^{-1} $year^{-1}$. From exclosure studies, Witt *et al*. (2011) calculated 0.22(0.02) Mg ha^{-1} $year^{-1}$ in biomass for rehabilitation of the mulga lands. Those three rates are comparable with that derived for mesquite (*Prosopis* spp.) thickening in Texas: 0.52(±0.12) Mg ha^{-1} $year^{-1}$ in SOC and aboveground vegetation (Archer et al., 2004). Afforestation of rangeland through technology to access groundwater under hardpan can achieve net sequestration of at least 1.1 Mg ha^{-1} $year^{-1}$ (Yamada et al., 2003).

The C sequestration flux of 0.36(±0.18) Mg ha^{-1} $year^{-1}$ calculated here for QLD regrowth biomass is higher than for rangeland rehabilitation and the exclosure studies, but lower than for thickening, and closest to the Garnaut (2008) estimate. Juvenile regrowth may have mature live root stock from before deforestation (e.g.

brigalow (*Acacia harpophylla*) has lignotubers). Therefore the growth rate for regrowth should be faster than where colonisation (or re-colonisation) is required, such as for thickening or rehabilitation.

Arrest of further deforestation and re-clearing has been termed 'forgone development' and a monetary cost allocated (Davidson et al., 2006). However the cost of C efflux accompanying further deforestation was not included. Also, prior to the high deforestation rates of the 1990s in Queensland, avoidance of deforestation of eucalypt forest was recommended by the Queensland Department of Primary Industries as it precipitates woody thickening (Turner and Beeston, 1978; Pressland, 1984). Although the purpose of deforestation is to provide improved grazing pasture, the benefits are not always perpetual; for example: (a) higher productivity only lasting for a few decades; (b) phosphorous levels decreasing to pre-deforestation levels within 10 years; (c) some nutrients increasing while others decreasing, and (d) sodicity increasing, due to reduced activity of deep-set tree roots (Sangha et al., 2005; Kaur et al., 2006; Kaur et al., 2007; Radford et al., 2007).

Raison *et al*. (2009) estimated that a C emission of 1.9 Tg year^{-1} could be avoided by not re-clearing regrowth on freehold land in QLD. Moreover, allowing regrowth to mature could make a significant contribution to Australia's climate change mitigation effort: the 8.2 (±4.1) Tg year^{-1} (calculated here) would offset either: (a) Australia's fugitive emissions, or (b) ~50% of Australia's agriculture emissions reported in the national inventory (as of mid 2011, DCCEE, 2011). There would also be biodiversity, soil rehabilitation and downstream water-quality benefits. Recovery of forest cover, through managed reforestation, natural regrowth, or woody-thickening on degraded land, is however, likely to reduce pasture cover (at least for several decades) because tree basal area is negatively correlated with herbaceous cover (Beale, 1973). (Although some tree and shrub cover is beneficial for grass growth on more-xeric sandy soils Daly and Hodgkinson (1996)). A middle-ground scenario with proportions of both herbage cover and tree cover can provide a compromise. Clumped trees (rather than isolated trees) and spatial variability in tree cover (for a given basal area) is beneficial as it can greatly increase soil nutrients,

SOC and pasture production (Scanlan, 2002; Eldridge and Wong, 2005). In terms of biomass carbon, optimization is also possible through the cover provided by a few mature paddock trees, which can have high biomass, but less canopy cover than numerous saplings, though with higher canopy cover for the same basal area. This is similar to the effect of shrub cover limiting grass growth more than does tree cover (Daly and Hodgkinson, 1996). However, this balance of mature trees and grass requires several decades for self-thinning and maturity of the regrowth (or woody thickening), which can be stymied under poor environmental conditions and by browsing. Meanwhile, reduced pasture production would prevail. Also, once an exotic grass is established, especially in nutrient-rich sites, then initiating reforestation may be problematic (Semple and Koen, 2003; Prober et al., 2011).

9.5.2.2 Sequestration in soil

For rangeland rehabilitation by destocking, based on average literature values of sequestration in soils, Gifford and McIvor (2009) estimated SOC sequestration flux at 0.07 Mg ha^{-1} $year^{-1}$ and 0.14 Mg ha^{-1} $year^{-1}$ for degraded mulga lands and all Australian rangeland, respectively (with an error margin of a factor of 10). That sequestration was to occur over 40 years. From exclosure experiments Witt *et al*. (2011) calculated SOC sequestration of 0.049 Mg ha^{-1} $year^{-1}$ to 0.3 m, for rehabilitation of the mulga lands. From modelling of destocking Harper *et al*. (2007) estimated SOC sequestration at 0.0091(±0.0061) Mg ha^{-1} $year^{-1}$ over five years. Those studies did not include landscape-scale effects such as ongoing riparian erosion (including streambank erosion, gullying, sheet erosion and grass removal, exacerbated by periods of drought Condon (1972)), and therefore did not represent the net SOC flux at the landscape-scale. The estimate of Harper *et al*. (2007) was most likely lower as it was a very short-term flux and it may have increased with time, following a logistic curve.

The slower flux of SOC arises from the longer average half-life of SOC than that of dead biomass. Significant contributions to SOC come from coarse and fine root

turnover (Lorenz et al., 2007; Kirschbaum et al., 2008)— these are not replenished after decomposition following deforestation. Due to the long half-lives of some SOC pools, for SOC to reach potential at the site-level (i.e. maximum SOC storage), several centuries are required (Hibbard et al., 2003; Chapter 8). The SOC sequestration component of rehabilitation modelled here used the time-dependent output from the modelling of woody-thickening by Hibbard et al. (2003), which had a logistic curve (starting slowly, faster in mid-range and slowing towards 2000 years). Their average rate for SOC sequestration amongst woody vegetation was 0.15(±0.04) Mg ha^{-1} $year^{-1}$, which is most similar to the Australian rangeland rate of Gifford and McIvor (2009).

Both emission (from erosion) and sequestration (from woody vegetation) were included in the calculation of ΔSOC with rehabilitation, therefore yielding a time-variable rate for net SOC flux. Upon destocking it is unlikely that instances of severe riparian erosion would be curtailed without active intervention and aeolian erosion may only cease after seed germination following rain, possibly combined with gradual litter accumulation. Thus, after onset of rehabilitation nation-wide, both emission and sequestration would overlap. In that situation the calculated net effect of a reduced C emission down to 0.36(±0.32) Pg, rather than sequestration per se, is the flux relevant national climate change mitigation efforts.

Disturbances that cause tree death can liberate otherwise stable SOC, through the 'priming effect' thereby producing additional carbon efflux (Dijkstra and Cheng, 2007; Fontaine et al., 2007). The combination of influences means that the SOC stocks can decline notably through frequent deforestation or timber harvesting. SOC dynamics are complicated and ΔSOC accompanying change in biomass requires comprehensive experimentation, especially close to mature trees (e.g. Throop and Archer, 2008) or to deep levels (e.g. Young et al., 2005), where deep-rooted plants contribute to SOC from root-turnover, and to where decomposition residues may accumulate through eluviation in coarse-textured soils).

9.5.3 Climate change effects

The often-perceived paradigm of an increase in biomass with anthropogenic global change (including climate change, land-use change and atmospheric pollution) is based mostly on: (a) increased CO_2 causing faster photosynthesis, (b) increased CO_2 causing increased water use efficiency, (c) possible nitrogen fertilisation from pollution, and (d) in some locations, longer growing season due to warming. There is suggestion that the anthropogenic increase in CO_2 has, at least until recently, increased biomass in Australia (e.g. Howden et al., 2001; Berry and Roderick, 2006). Increased biomass in the Australian rangelands due to the CO_2 and N fertilisation effects of further global change has also been modelled (e.g. Carter et al., 2004), but not measured.

The balanced chemical equation below represents the formation and disassociation of cellulose (the main constituent by weight of terrestrial vegetative biomass). The equation, with an equilibria sign to indicate that it goes in forwards and reverse directions, is written without showing the intermediaries (such as glucose, soil microbial pathways, and methane under anaerobic decomposition) for visualization purposes.

$$n6CO_2 + n5H_2O + \text{Energy} \underset{\text{Combustion or Decomposition}}{\overset{\text{Photosynthesis or Catalysis}}{\rightleftharpoons}} (C_6H_{10}O_5)_n + n6O_2\uparrow \quad \text{Eq9-1}$$

For a vascular plant, the effects of the reagents (e.g. CO_2, H_2O) are linked through a range of secondary processes dependent on environmental conditions, which may act neutrally, positively, or as environmental stressors, on the forward and reverse reactions; such that an increase in a reagent (e.g. CO_2) does not produce the result indicated by the equation alone. For example, in one CO_2 fertilisation experiment, the N uptake by grass from the more-recalcitrant SOC was increased, leaving more-labile SOC, and net shorter half-lives of ecosystem carbon (Gill et al., 2006). Similarly, although the formula used in the present work for determining change in

SOC as a function of temperature and rainfall (Bird et al., 2002) had an R^2 of 0.75, it was not tested for use in rapid temporal change. The substitution of time for space may be true to some degree but is a simplification of the ecosystem dynamics of climate change, which require further consideration.

Some of the secondary processes have been described for rangeland vegetation (e.g. Walker and Steffen, 1993; Polley et al., 2011). The more abiotic ecological factors associated with climate change, such as: (a) drought stress (longer and hotter droughts), which limits the forward reaction in Eq9-1; and (b) increased fire frequency, which favours the backwards reaction— both of which have been forecast for Australia by 2100 (Pitman et al., 2007; Pitman and Perkins, 2008; McAlpine et al., 2009; Williams et al., 2009; Liu et al., 2010)— are usually not included in the more-singular "process based" models (such as modelling the effects of increase in only CO_2 and N).

Potential evaporation is forecast to increase for most of Australia's rangeland, which could be expected to affect soil moisture, surface runoff, water supply and water demand (i.e. water vapour deficit) (e.g. Bates et al., 2008). An example of modelling of intermediate complexity— including changes in CO_2, temperature, rainfall and drought, but not including change in fire regime— revealed for northeast America, a variety of spatially dependent responses, with repeated episodes of dieback and significantly reduced biomass, but in some locations eventual forest growth upon quadrupling of CO_2 concentration (Solomon, 1986). Such a high CO_2 concentration would induce positive feedback, and temperature is rising faster than in the modelling by (Solomon, 1986), which accommodated ecological adaptation of local taxa.

Determination of increased biomass with anthropogenic climate change has been based on a range of models or observations, such as: (a) lower forest biomass during glacial periods (with their inherently low CO_2 concentrations, (e.g. Street-Perrott et al., 1997)), (b) high photosynthesis rates in some CO_2 enrichment experiments (Saxe et al., 1998), (c) forests where water was not strongly limiting (Boisvenue and

Running, 2006), and (d) change in vegetation indices in remotely sensed imagery (e.g. Piao et al., 2011b).

Some such forecasts of increased biomass with climate change may be discounted, because (a) the causes and possibly some characteristics of anthropogenically-induced climate change are different to those of previous glacial retreat, (b) measurements using flux towers alone would neglect efflux of [aqueous] dissolved organic carbon and [airborne] volatile aerosols (Beringer et al., 2007) and thereby overestimate net ecosystem productivity, and (c) some measured biomass increase was due to inherent forest recovery from past disturbance (Heath et al., 1993; Cook et al., 2005; Chave et al., 2008; Muller-Landau, 2009).

Moreover there is ample evidence for a decrease in biomass with ongoing anthropogenic climate change, albeit initiating at different times in different locations. Damaging influences of continued anthropogenic climate change upon several extant ecosystems globally have been modelled and observed (Chapter 1). Drought effects will worsen in many regions that are already water-limited, and they have been observed in areas previously not considered as being water limited (e.g. Allen et al., 2009). Such negative aspects are likely to detract from sustainability of existing forests and shrublands, and their ability to store carbon; and will thereby induce net carbon efflux (i.e. positive feedback to climate change, cf. Houghton (1997), Kashian *et al.* (2006)).

Some specific examples of reduced biomass [while under the influence of increasing CO_2, N and temperature] are: (a) since 1990 decreased forest cover in unmanaged areas and reduced recovery in thinned areas (Wallace et al., 2009) resulting from reduced rainfall in southwest Western Australia, since c. 1975 due in a significant part to anthropogenic climate change (Bates et al., 2008; Pitman and Perkins, 2008) and; (b) reduced growth of forests in Eurasia during the growing season since c. 1997, also concomitant with decreased rainfall (Piao et al., 2011b). Notably in the latter example, prior to 1997 there was increased growth, possibly due to climate change affects, before the affects became too stressful (and swinging the chemical

equation for cellulose production more to the reverse direction than the forward). Thus, for some locations, the 'honeymoon period' of anthropogenic climate change has passed.

Carnicer *et al.* (2011) noted that the observed negative effects (i.e. constituting reduced growth or reduced biomass) of climate change, were more pronounced in more xeric localities. The reduced soil moisture (from evapo-transpiration affects) will reduce resistance to erosion, thereby decreasing productivity (Thorne et al., 2005; Eldridge et al., 2011b). That suggests that the dryer parts of Australia are more likely to be affected. Additionally, vulnerability to drought increases with land degradation and [human] population increase (Pickup, 1998)— which is relevant to the majority of Australian rangeland.

Net primary productivity may increase with increase in precipitation, however soil has high sensitivity to temperature increase— increasing emissions from SOC and decreasing net ecosystem productivity (Peng et al., 1998; Xuyang et al., 2008). Where SOC is forecast to decrease, there could be net ecosystem C sequestration if increase in precipitation is sufficient (e.g. Doherty et al., 2009), however no increase in precipitation was consistently forecast for Australian rangelands amongst the different models tested here.

Positive feedback will increasingly augment climate change (Scholze et al., 2006; Denman et al., 2007; Piao et al., 2011b). One mechanism of positive feedback is that, increased fire frequency (such as forecast for Australia over the next century) generally changes vegetation types to those with more dispersed tree coverage, enlargement of tree hollows, increased proportion of more fire-tolerant species, and increased grass cover, i.e. changing mature or mesic forest to savannah or grassland— a more-fire-prone ecosystem (Cloudsley-Thompson, 1975; Cook et al., 2005; Pittock, 2009) which has a lower ecosystem carbon. That process is the likely pathway of the ΔSOC calculated here. Increased frequency of fire initially acts to remove young tree recruits and in the longer term removes the more mature trees (which become more vulnerable to fire with age) (Liedloff et al., 2001). However,

under conditions of minimally intense fires, drought tolerance of some shrub species (such as phreatophytes) may allow dominance over some grasses that are without significant below-ground storage components (White et al., 2009). Other positive feedback mechanisms are: (a) reduced site-quality (e.g. through soil nutrient loss and reduced clay content) from more-frequent fire and increased erosion (e.g. McIntosh et al., 2005; Gough et al., 2007b; Williams et al., 2009), (b) lower local rainfall due to reduced forest biomass (Li et al., 2000; McAlpine et al., 2009); and (c) increased efflux from increased SOC decomposition (Kirschbaum, 2004).

Carter *et al.* (2004) forecast an increase in rangeland soil organic carbon (SOC), to 0.3 m depth, of ~4% up to 2070, based on CO_2 fertilisation of vegetation growth, change in rainfall and temperature, but ignoring the increased fire frequency and the increased drought severity forecast for Australia (e.g. Williams et al., 2009). That modelling forecasted 'small' losses in woody biomass under drier conditions. Based on CO_2 fertilisation alone (without change in temperature or rainfall) they found a 5% increase in carbon and concluded that N availability was limiting. Phosphorous is also notably limiting over much of Australia and it is unlikely to be bolstered by global change. With more comprehensive modelling to 2100 (although still without fire and drought), the forecast change in SOC for Australia was 0.6% and -6.4%, for low- and high-emission climate change scenarios respectively (Grace et al., 2006), with highest losses in the mid- to northern-latitude rangelands. Also with comprehensive modelling (although without modelling change in drought or fire) an increase in SOC was forecast for Patagonian shrubland, peaking around 2050 but decreasing thereafter under a high-emission scenario and no net gain thereafter for the low-emission scenario (Carrera et al., 2007).

Another model of vegetation change, based on temperature and rainfall change with climate change (as was the modelling of ΔSOC), forecast an extensive expansion of the arid interior of Australia by 2070, at the expense of eucalypt forests (Adams-Hosking et al., 2011). A reduction in the northern rangeland forests by desert expansion has been forecast (Zeng and Yoon, 2009). With SOC generally being correlated with woody biomass (Jackson and Ash, 1998; Woomer et al., 2004; Wynn

et al., 2006), these changes would eventually incur SOC efflux. McKeon *et al*. (1998) forecast a SOC efflux of 50% from Queensland rangelands, from modelling of global change effects (although without increase in fire), accompanying a temperature increase of 2.7°C, which is likely over much of Australia by 2100 under a high emission scenario (Pitman and Perkins, 2008). The SOC efflux figure of 32(9)% (Table 9-3.b) for QLD for the coming century is of a similar order of magnitude to their value, within likely error margins. More generally, the magnitude of change in SOC modelled in the present work is within expected ranges when major ecosystem effects (such as fire and drought) are considered, although the timing of the flux must also be a major consideration.

The method of modelling the influence of climate change on SOC in the present work was based on differences between ecosystems, and dependent on mean annual temperature and rainfall and thereby it inadvertently also reflected differences in ecosystem characteristics such as fire and drought regimes (*cf* Chapter 2). Fire and drought are some are forecast to have increasingly negative impacts on much of Australian carbon stocks. The link between ecosystem change through fire has been applied previously in climate change modelling (e.g. Krawchuk et al., 2009). The SOC measured in development of the relationship of Bird *et al.* (2002) was inherently dependent on biomass (section 7.5), and consequently the modelled ΔSOC (-38(12)% to 0.3 m) inherently reflected change in biomass. However, decrease in SOC is generally significantly delayed after decrease in biomass, due in part to the surge of decomposing biomass after tree mortality, and to the long half-life of some major soil pools (e.g. from 250–400 years, Liao *et al*. (2006a); Neff *et al*. (2009)). Thus, rather than the calculated 38(12)% drop in SOC happening over a century, it may well take several centuries. The forecast 4% increase in SOC from Carter *et al.* (2004) ignored some aspects of anthropogenic climate change that are shown to be currently active. That increase will mostly likely be reduced. Also, considering the observed, globally widespread decrease in biomass and in biomass growth, if there is any further biomass increase due to climate change in Australian rangeland then it may only happen in the very near future. Consequently, any increase in SOC due to increase in biomass would be indistinguishable against the drop of 38(12)% in the

longer term. That calculated efflux of SOC is not likely to occur in one century, but it can be stated as a decrease likely to occur over several centuries resulting from the change in biomass occurring from 2000–2100. Additionally, if the ecosystems are unable to adapt then the forecasted ΔSOC may be a best-case scenario, with higher efflux likely.

The estimated total SOC efflux for Australia due to the climate change impact on rangeland ecosystems to 2100 was 1.8(0.6) Pg (a significant quantity). It was not possible to say which climate change model or IPCC scenario was the most accurate. The CSIRO Mk3.5 model consistently forecasted higher emissions: 55% and 72% for A1B and A1FI respectively. If that model and scenarios are proven to be the most accurate then emissions from SOC are likely to be higher than the average. Current trends are above the IPCC SRES scenario A1FI (Pittock, 2009), i.e. influences will be stronger than the average used here, in which case the SOC efflux from Australian rangelands could well be in the order of 1–2 Pg by 2100.

9.5.4 Sequestration: accounting, threats and enhancement

For successful rangeland C sequestration four avenues have been suggested (McKeon et al., 1992): (a) a more reliable estimate of the expanse of degradation, (b) monitoring the changes in carbon stock with time, (c) demonstration of the attainable sequestration, and (d) logistics for decreasing stock numbers. They also described a variety of effects of future climate change, which were mostly negative for both livestock and carbon sequestration. Gifford and McIvor (2009) concluded that before reliable estimates of sequestration can be made much more information was required on SOC dynamics, such as accompanying land degradation and rehabilitation. Calculations here showed that uncertainty in the root:shoot ratio (i.e. in root biomass) was a major contributor to uncertainty in emissions and sequestration. The correlation between long-term biomass and SOC (Jackson and Ash, 1998; Wynn et al., 2006) would indicate accompanying uncertainty for ΔSOC.

For sustainable management in the rangelands more effort is needed in assessing the extent, location and severity of land degradation (CIE, 2000) (these being sites for potential sequestration). Historically, rangeland research has focussed on pasture production for livestock rather than tree biomass or riparian zone condition, or C fluxes. State-held, detailed data on rangeland condition (at the paddock and property-levels), watering point developments, and on livestock stocking numbers, are of highly restricted access (even between government departments), and even for non-freehold properties leased from the government. Assessing rangeland degradation necessitates remote-sensing (e.g. Wallace et al., 2006) however its adoption requires: (a) better coordination between the government departments (e.g. lease administration, conservation, livestock trade, and infrastructure development), (b) innovation to find spectral band combinations applicable to high spatial variability in terrain and vegetation, and with minimal ground-truthing (e.g. Dean, 2005), (c) purchasing imagery of sufficient temporal and spatial resolution, and (d) resources for sufficient ground-truthing. In a review of Australian rangeland studies Foran (2007) estimated that it would be ~2030 before adequate spatial methods were applied nationwide.

The difference between the estimates for C emissions from livestock of 8.2 and 13 Tg $year^{-1}$ (Robertson, 2003; Cook et al., 2010) (with the latter including an undisclosed amount from outside the commercial rangelands) indicates another major source of uncertainty in annual emissions (here equal to 15% of 'Total2'.) This is augmented by the lack of data on livestock numbers, with restrictions on data access even between government departments (personal communication, Greg O'Reilly, NT government, 2005).

Complexities arise for carbon accounting in rangelands, from: (a) high 3D spatial heterogeneity of SOC stocks (e.g. higher under woody vegetation and at deeper levels where phreatophytes access deep-set ground water), (b) long timelines for change, (c) multiple land uses, (d) contribution from aeolian, surface water, and groundwater transport of carbon, and (e) probable fire increase accompanying climate change and population increase. SOC may decrease due to high rates of fuel-

wood extraction (Kaul et al., 2009). The Australian rangeland forests are used also for timber extraction, both for firewood collection and commercial forestry on private estates, where the extraction volumes are more indeterminate than in State forests (Henry et al., 2002) and hence so are the impact on C stocks. For example, sandalwood (*Santalum spicatum*) has been extracted (primarily for its aromatic oil) from Western Australian since *c*. 1850, but is now sourced from a reduced area of ~50 Mha of rangeland. From 1850–1900 approximately 0.175 Tg were extracted, roughly equal to the stock available in 1970 (Kealley, 1991). (This is however only around a thousandth of the biomass lost in QLD deforestation.) From data in (Kealley, 1991) and (Kealley, 1989), from 1984–2011, the harvestable stock was reduced by 36%. Sustainability of extraction is in doubt due to poor regeneration compared with the extraction rate (Kealley, 1991). Due to sandalwood's sensitivity to fire, grazing by livestock and feral herbivores, diminished stocks and the need for successive years of good rainfall for regeneration (Kealley, 1991); climate change is expected to impact negatively on sandalwood sustainability, and a long-term impact on both biomass and SOC to some degree would be expected.

Buffel grass is highly prized by many pastoralists for livestock production. However, due to its impact on biodiversity (Fairfax and Fensham, 2000; Clarke et al., 2005; Jackson, 2005; Smyth et al., 2009; McDonald and McPherson, 2011), and on fire regimes (discussed below), and its invasive capacity, it is a declared pest plant or major environmental weed in Australia, with the potential to colonise over half of the mainland (e.g. Grice and Martin, 2006; Friedel et al., 2008; Australian Weeds Committee, 2016). Increased grazing pressure, deforestation, and thinning, are linked to colonisation of Australian commercial rangeland by buffel grass (Ludwig and Tongway, 2002; Eyre et al., 2009); and encroachment is facilitated by native grasses' previous lack of selective pressure from large mammalian grazers— presenting a point of advantage for buffel grass under livestock grazing (Hodgkinson et al., 1989; Ludwig and Tongway, 2002; Foxcroft et al., 2010).

Buffel grass infestations can reduce ecosystem biomass and SOC by attracting grazing pressure, depriving shrubs of moisture, and increasing the area burnt and fire

intensity during wildfire (e.g. causing increased mortality of woody vegetation) (Ibarra-Flores et al., 1999; Williams and Baruch, 2000; Clarke et al., 2005; King et al., 2008; Miller et al., 2010). For example, from modelling fuel loads and fire histories in central Australia King et al. (2008) found that with buffel grass, for a fire of moderate size (500 ha), the frequency of occurrence increased by 40%, and for fires of moderate frequency, the area burnt increased five-fold, but with greater proportional increases for larger, less-frequent fires.

The mechanisms of buffel grass's fire promotion and resilience to fire include: (a) maintaining high loads of flammable biomass, (b) more intense fires than to which native ecosystems are accustomed (McDonald and McPherson, 2011), (c) a relatively high, below-ground component that quickly resprouts after fire (Ward et al., 2006; Rao and Allen, 2010), and (d) a high surface-to-volume ratio compared with woody vegetation (increasing flammability). Also, anthropogenic nitrogen fertilisation can preferentially increase growth of invasive grasses over some native grasses, to the extent of greatly increasing fire risk (Rao and Allen, 2010). A cycle of invasive-grass, fire, grassland-dominance establishes, to the detriment of woody biomass (e.g. Westoby et al., 1989; Búrquez-Montijo et al., 2002). This will also contribute to lower SOC, as woody biomass is correlated with SOC (including for woody-thickening (Eldridge et al., 2011a)). Modelled increase in grass growth due to CO_2 fertilisation (even without buffel grass), and the accompanying increase in fire, has been shown to reduce carbon stocks in grazed, timbered rangelands (Howden et al., 2001).

As buffel grass congregates in the [moister] riparian zones (Figure 9-6), and under shrubs and CWD, i.e. in favourable microsites (McDonald and McPherson, 2011)— which is where the rangeland woody biomass is often concentrated (Figure 9-5, 9-6)— then it constitutes a threat to rangeland C stocks. The increased fire threat also prevails in remnant woodland that neighbours deforested areas colonised by buffel, as the higher wind speed in the latter promotes faster fire spread (Freifelder et al., 1996).

As fire is expected to increase in severity and frequency with climate change in Australian rangelands, then buffel grass presents a synergistic, increased danger of C efflux. Thus it will also contribute to climate change positive feedback. It may also spread further south via the extended aridity of the higher emission scenarios of climate change (Steel et al., 2008).

Buffel maintains biomass and spreads after destocking (D'Antonio and Vitousek, 1992). Possible reasons for the higher biomass of buffel that was observed inside livestock exclosures in WA are: (a) minimal grazing pressure in the exclosures, and (b) additional soil and nutrients blown in from the nearby grazed area and accumulation under small plants and shrubs (Spooner et al., 2002). The exclosures were isolated from fire by large expanses of heavily grazed land [that would not carry fire well]. However, if whole properties were destocked (e.g. for financial carbon sequestration schemes) then they would not be isolated from ignition sources and spread of fire, and would consequently be at increased fire risk after high accumulation of buffel grass. Reduced grass fuel loads accompanying rangeland commercialization have been implicated in woody-thickening (e.g. James et al., 1999). Similarly, through its increased fire effects, high grass biomass can reduce woody biomass in the absence of grazing (e.g. Werner, 2005). In the long-term that higher fire risk, could offset carbon gains from large-scale destocking, and thereby limit the fruitfulness of such ventures aimed at carbon sequestration. If properties were destocked for carbon sequestration, it may be necessary to retain some artificial watering points, for at least the grazing pressure exerted by native macropods.

Feral herbivores (e.g. camels, goats, pigs, buffalo and horses etc.) may not be constrained by destocking, standard fencing, nor decommissioning of artificial water points if springs etc. remained accessible. Consequently ferals, which can contribute to land degradation, would thereby continue to detract from rangeland biomass and SOC in some locations. Fencing of natural watering points would threaten the livelihood of larger native animals and therefore possibly of native ecosystem functions.

Reducing grazing pressure alone, especially if feral herbivores are present, is insufficient to restore native grasses and in some cases trees, and exotic grasses must also be controlled (Semple and Koen, 2003; Allcock and Hik, 2004). Methods of buffel management include control of dominance, eradication, reduction of spread, and suppression (Jackson, 2004; Franklin et al., 2006; Friedel et al., 2008; Brenner, 2010). Once tractable solutions for the buffel grass and feral animal issues are found, especially where they are likely to have significant influence over carbon stocks such as in riparian areas, then significant emissions from rangelands could be curtailed and sequestration encouraged. The specific C efflux from riparian erosion alone, currently has high uncertainty: a priority is to quantify its spatial distribution and intensity, for both carbon accounting and for monitoring rangeland condition. Without destocking and widespread land rehabilitation, a major source of C emissions can still be addressed by limiting riparian land degradation (being regions of higher C concentration). Mechanisms for restoring riparian ecosystems are discussed in Bainbridge (2008) and they begin with problem-area isolation. That reinforces the importance of directly addressing some major issues described in this study: (a) the juxtaposition of artificial watering points and riparian areas, (b) the erosion foci of vehicular or livestock crossings of riparian areas, (c) buffel grass infestations in riparian zones.

One management alternative, that would simultaneously promote reduced deforestation for grazing, rangeland degradation, and livestock emissions, while engendering rangeland sequestration through forest regrowth and rehabilitation, would be to substitute the management of rangelands for protein production from livestock [for human consumption], by *in vitro* protein production (e.g. Edelman et al., 2005; Hopkins and Dacey, 2008). That option would however require a major administrative intervention.

In summary, biomass and soil carbon are both major pools in rangeland carbon accounting, although the latter has a longer timescale. Livestock methane is a significant short-term contributor. The potentially forested rangelands govern the major C stocks and fluxes in both biomass and soil carbon. The fastest C

sequestration option, QLD regrowth of 0.79(±0.40) Pg, is less than half of the emission from SOC due to the influences of future climate change (1.8(0.6) Pg). Rangeland rehabilitation will only be possible by attention to major threats such as riparian erosion and buffel grass. The emission-limiting and sequestration options presented here were made independent of future climate change effects and consequently their quantitative values are likely to vary from those given here, towards reduced sequestration capability. There is a prevalent lack of precision in rangeland C fluxes but the knowledge base will slowly improve through comprehensive remote-sensing and experimentation, especially long-term studies of biomass and very long-term studies for SOC, through deep sampling near woody vegetation, and corresponding modelling of carbon dynamics. Additionally, increased transfer of information on factors affecting rangeland condition and the effects, are required between government departments and from there to the scientific community.

Chapter 10 Locating optimal carbon sequestration in arid and semiarid rangelands

This chapter is adapted from a published paper: Dean et al. (2015)

10.1 Abstract

Destocking degraded rangeland can potentially help climate change mitigation by re-sequestering emitted carbon. Broad-scale implementation has been limited by uncertainties in the magnitude, duration and location of sequestration and the profitability relative to the existing grazing land use. This chapter employs a novel methodology to assess potential rangeland sequestration and its profitability, using 31 Mha of rangeland in New South Wales, Australia as a case-study. This approach combines remotely sensed data and modelled estimates of various components. Remotely sensed, synthetic aperture radar data were used to determine woody biomass of minimally degraded forest (benchmarks) and neighbouring more-degraded forest, followed by sequestration modelling using non-linear growth rates based on woody thickening and slow-growing plantations, scaled to the benchmarks. Livestock concentration and livestock-based farm profits were modelled. The sequestration and grazing net profits were compared, for a carbon price of AUD$10 Mg^{-1} CO_2-e, at different growth stages for different levels of forest attrition. It was found that broad-scale destocking with subsequent C re-sequestration was initially unprofitable compared with grazing. However, after 50 years, with full costing of C emissions, the returns were similar for the two alternatives of continued grazing or re-sequestration, for areas with biomass below benchmark levels. Reforestation of recently deforested land represents the most profitable option with profitability increasing with growth rate. Emissions of soil organic carbon, set in motion by climate change over the next century, were calculated to be the largest of all sources.

Emissions from biomass, induced by climate change, will be higher where vegetation cannot adapt. The secondary effects of climate change will reduce re-sequestration and grazing profits, possibly limiting the carbon stored by re-sequestration projects.

10.2 Introduction

Rangelands supporting commercial livestock grazing are contested ground in which production of meat and other animal products for the increasing human population, nature conservation and conserving or replenishing carbon stocks increasingly compete for space (e.g. Glenn et al., 1993; Schuman et al., 2002; Reid et al., 2004; Dutilly-Diane et al., 2006; Khan and Hanjra, 2009; Janzen, 2011). These rangelands have commonly experienced net vegetation and soil loss (e.g. Allen, 1983; Fanning, 1999; Zucca et al., 2010; Dotterweich, 2013) corresponding to net C (carbon) emission. Reversal of land degradation linked to that carbon emission process can theoretically replenish the lost C (Howden et al., 1991; McKeon et al., 1992; Glenn et al., 1993; Walker and Steffen, 1993; Henry et al., 2002). The refilling of that depleted carbon stock (henceforth termed re-sequestration) contrasts with sequestration projects storing C in a form or location different to its origin (e.g. afforestation or power-station carbon capture and storage). Uncertainties in the potential magnitude, duration, location and profitability of carbon re-sequestration projects, have limited their implementation.

Rangeland emissions can be lessened by reduced deforestation, protection and enhancement of soil organic carbon (SOC), and by reforestation (Henry et al., 2002). Reforestation can be intensively managed (e.g. plantings), or passive/'natural' (e.g. Rey Benayas et al., 2007; Grainger, 2009) by allowing woody thickening (i.e. infill, Rackham (1998)) and regrowth to mature. The passive type is considered here, though managed reforestation can be used if finances permit. SOC stocks are generally positively correlated with aboveground biomass (Jackson and Ash, 1998; Harms et al., 2005; Young et al., 2005; Wynn et al., 2006), being primarily derived from root turnover and litterfall. This relationship accounts for some of the decline in

SOC stocks with vegetation attrition (Chapter 8), the remainder being through erosion pathways (Chapter 9). Magnitudes of change in soil organic carbon (ΔSOC) have higher uncertainty than associated changes in rangeland biomass (Henry et al., 2002). Consequently the present work focuses on biomass but with discussion of linked ΔSOC.

Rangeland C re-sequestration opportunities coincide with overgrazing or deforestation. Localised benchmarks of potential C stocks can be derived from remnant ecosystems or spatially dependent environmental variables (Greve et al., 2013). This equates to determining 'carbon carrying capacity' (Roxburgh et al., 2006b). The potential of plantations to replenish C on deforested semiarid to mesic arable land in southern Australia was estimated by Paterson and Bryan (2012). The work in this chapter is thematically similar, but it uses a finer spatial scale, simulates natural (autonomous, unmanaged) regrowth, allows a longer duration, and— to reflect the higher error margins in rangeland— calculations are at a coarser economic scale.

Remote-sensing calibrated by ground-truthing, or, more frequently, ground-based assessments alone, are employed in rangelands for regular land condition assessments and woody biomass monitoring. Adaptation of remote-sensing technology by land managers is slowly approaching a level suitable for routine operational usage over the large expanses for which it was originally intended (e.g. Graetz et al., 1976; Mackay and Zietsman, 1996; Ustin et al., 2009). Both LANDSAT and the Advanced Land Observing Satellite (ALOS) Phased Array L-band Synthetic Aperture Radar (PALSAR) sensor have proven applicability for aboveground biomass assessment of arid and semiarid open woodland (Armston et al., 2010; Lucas et al., 2010) with radar more sensitive to woody biomass and LANDSAT more sensitive to vertically projected foliage distribution (Armston et al., 2009; Danaher et al., 2010). PALSAR has proven applicability for carbon flux assessment in complex situations, though the basic radar data are often integrated with other data types, such as LiDAR or LANDSAT, or undergo more complex processing (e.g. Carreiras et al., 2012; He et al., 2012; Sarker et al., 2012). In the

present work pre-processed data from PALSAR is principally used, and results are compared with those from the lower resolution NOAA-AVHRR sensor.

Managed reforestation, including rehabilitation of degraded rangeland to enable woody regrowth where there has been substantial top soil loss, may require financial inputs (Spooner et al., 2002; Sparrow et al., 2003; Mengistu et al., 2005; Neff et al., 2005; Polglase et al., 2013). No financial inputs would be necessary if a low C sequestration rate, similar to that for passive reforestation of degraded and grassy areas by natural regrowth and 'woody thickening', can be applied.

The main aims in this chapter are to determine the most lucrative places in rangeland for C re-sequestration, and to develop a method for determining the C re-sequestration potential and rates for those areas. This is applied to rangeland in New South Wales (NSW) Australia, which is used largely for the generation of profit from grazing domestic livestock (henceforth termed commercial rangeland). An understanding of the relevance of the findings to rangeland outside of the study area is facilitated by a global climate and biome comparison. Options for avoidance of any carbon emissions leakage after destocking are discussed. A notional carbon price is used for comparative purposes, fully realising that there is a long way to go before markets for carbon and rules for accessing such markets gain widespread acceptance.

10.3 Methods

10.3.1 Terminology and definitions

The boundary of the Australian rangeland zone has been variously mapped (Donohue et al., 2005). The definitions and area of Australian forest and rangeland used in this chapter are those from Chapters 1 and 9.

The term 'forest degradation' is used to refer to both gradual attrition of a forest's woody biomass (whether through erosion from overgrazing, thinning, or reduced

water infiltration from weeds or loss of debris etc.) or immediate deforestation, such as in 'land clearing' and 'fence-line clearing'.

Most spatial data on the weights of dry biomass are available for aboveground only. For total tree carbon in biomass it is assumed that the root:shoot ratio is 1:3, i.e. 25% is below ground (Burrows et al., 2000; Henry et al., 2002). This is a conservative value compared with other literature values for arid and semiarid ecosystems (Mokany et al., 2006).

In the calculations in the present work dry biomass was considered to contain 50% carbon, by weight, as in Chapter 3. Carbon assay for semiarid *Acacia* species in Western Australia (sampled in a separate project) showed proportions of C close to 50%.

The value used for the 'price of carbon' was AUD$10 Mg^{-1} CO_2-e (CO_2 equivalent). It is close to the Climate Action Reserve's (USA) value of AUD$11.50, the European Union's ETS value of AUD$7.94, and China's Shenzhen value of AUD$8.02 (prices as of January 2014). The value of AUD$10 Mg^{-1} CO_2-e equates to a price for C of AUD$36.641 Mg^{-1}. Note that the 'carbon price' can vary depending on the market in which it is traded, the source of the carbon, and over time in response to both policy decisions and supply and demand. Carbon masses used herein are in terms of C, not CO_2-e (unless otherwise specified).

10.3.2 Study region and data processing

Supplementary information is in Appendix V. Sources of NSW data are in Table A-V-1.

All area calculations were performed in Albers Equal Area projection, as it provides minimal distortion over large areas (Steinward et al., 1995). Areas of commercial rangeland within NSW were determined from land-use data ('LanduseV1', Anni Blaxland Fuad, NSW Office of Environment and Heritage, personal communication,

2011) containing 150 land-use classes (Figure 10-1, Table A-V-2). Other countries and other Australian States with similar ranges of annual rainfall and average temperature (i.e. potentially similar climate), and similar biomes, to the study area in NSW were identified using GIS.

The NSW [commercial] rangeland lies within the western-most six Livestock Health and Pest Authority (LHPA) districts (Figure 10-2) and within the Australian defined rangeland zone. The area within that zone but to the east of the six LHPA districts contained a minimal amount of commercial rangeland (0.056 Mha) but a significant amount of deforested, non-woody pasture (1.2 Mha), and was ignored as it required including parameters for a seventh LHPA district 'Central North' which was mostly outside of the rangeland zone. More generally, the landuse type of grazing 'Volunteer, naturalised, native or improved pastures' (which is deforested land) with no regeneration of woody cover was not included as it generally consisted of undifferentiated 'native' or 'exotic' species, and could not be differentiated from intensive grazing, apart from the absence of irrigation, and as such it did not come under the definition of grazing native or semi-native vegetation (i.e. rangeland) (Figure 10-3). It occupied 6.50 Mha within the rangeland zone and 5.31 Mha within the six westernmost LHPA districts, and was distributed mostly in the higher rainfall region to the east.

Within the greater rangeland zone for NSW, values for environmental attributes, ecosystem-type (from Mitchell Landscapes data, described in e.g. Gibbons et al. (2009)) and extant SOC to 0.3 m (Barson et al., 2002) were assigned to 62,371 polygons of conterminous land-use type (using ESRI ArcMAP GIS), ranging from 1–780,000 ha, with an average size of 490 ha. Polygons smaller than 1 ha summed to <1% of the total area and were discarded. Where single polygons overlapped a range of environmental values, then either an area-weighted average or majority was assigned, for numeric and thematic data, respectively. NSW rainfall and temperature data were mean annual averages from 1961–1990, with a pixel size of 0.025º (~25 km) (Bureau of Meteorology). Spatial data on deforestation, as determined from

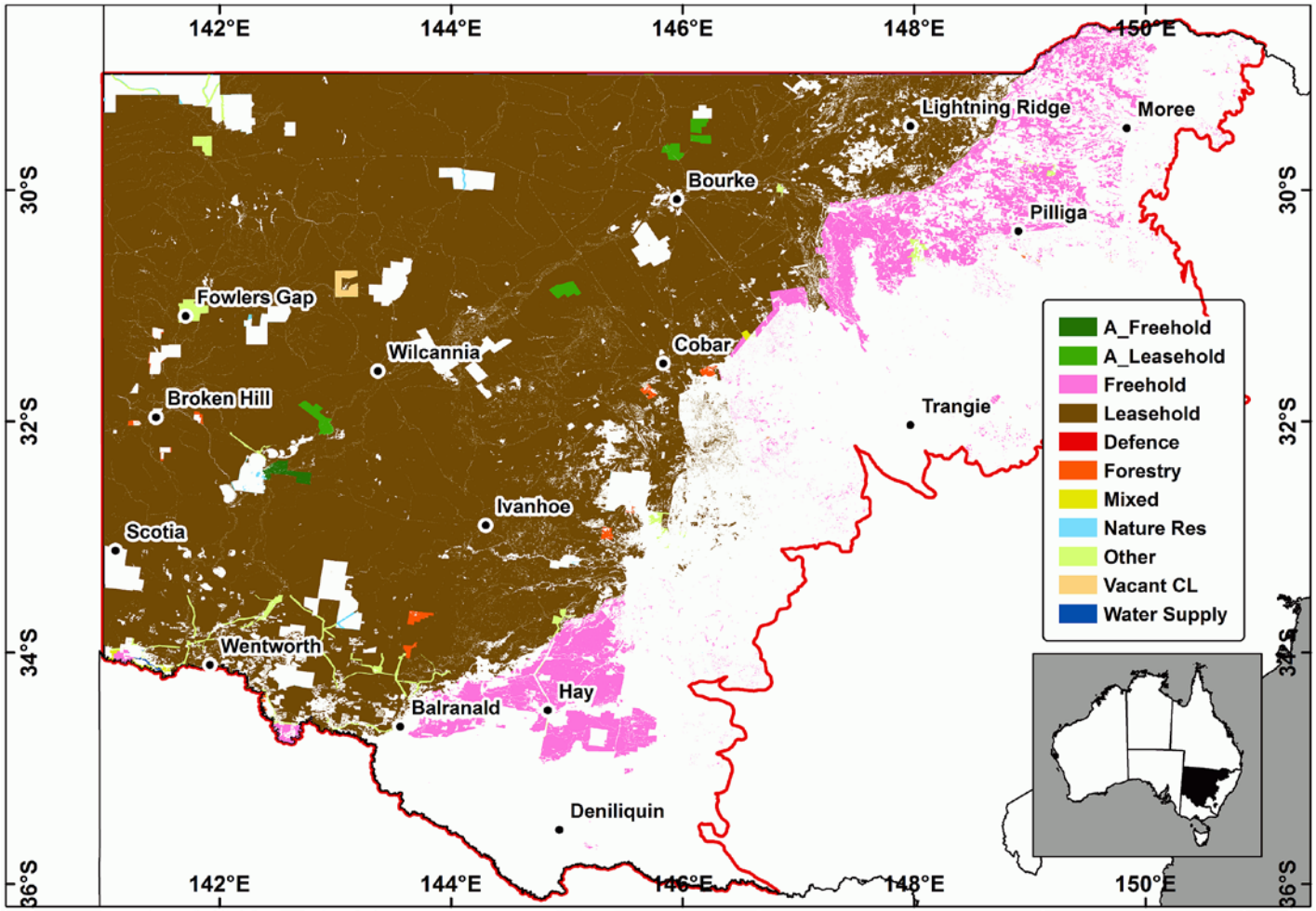


Figure 10-1. Distribution of land tenure in NSW rangeland. Rangeland zone= west of red line. Abbreviations: 'A'= Aboriginal, 'CL'= crown land, 'Res'= reserve, 'Mixed'= multiple-use public land. Geographic projection.

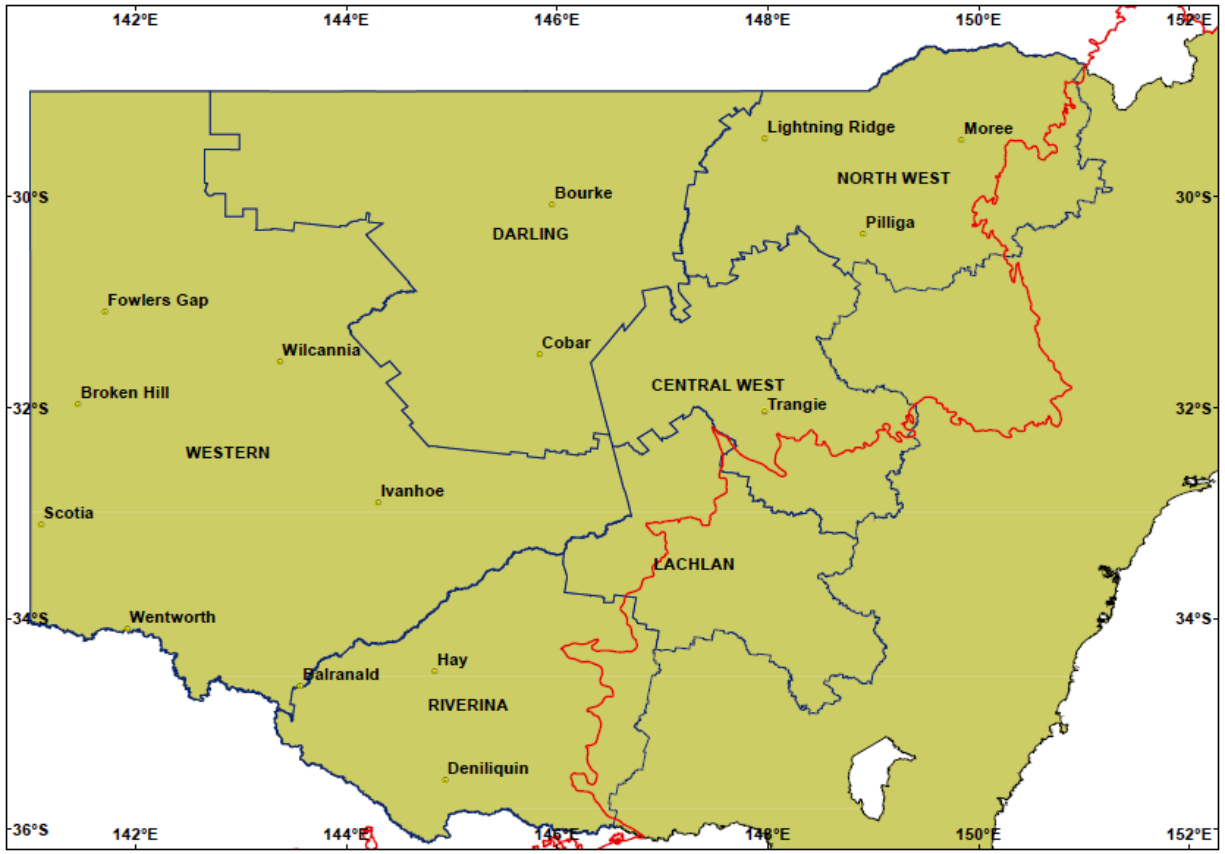


Figure 10-2. Australian defined rangeland zone (west of red line) overlaid on the Livestock Health and Pest Authority (LHPA) districts (blue lines). The study area was to the west of the red line and within the western-most six LHPA districts.

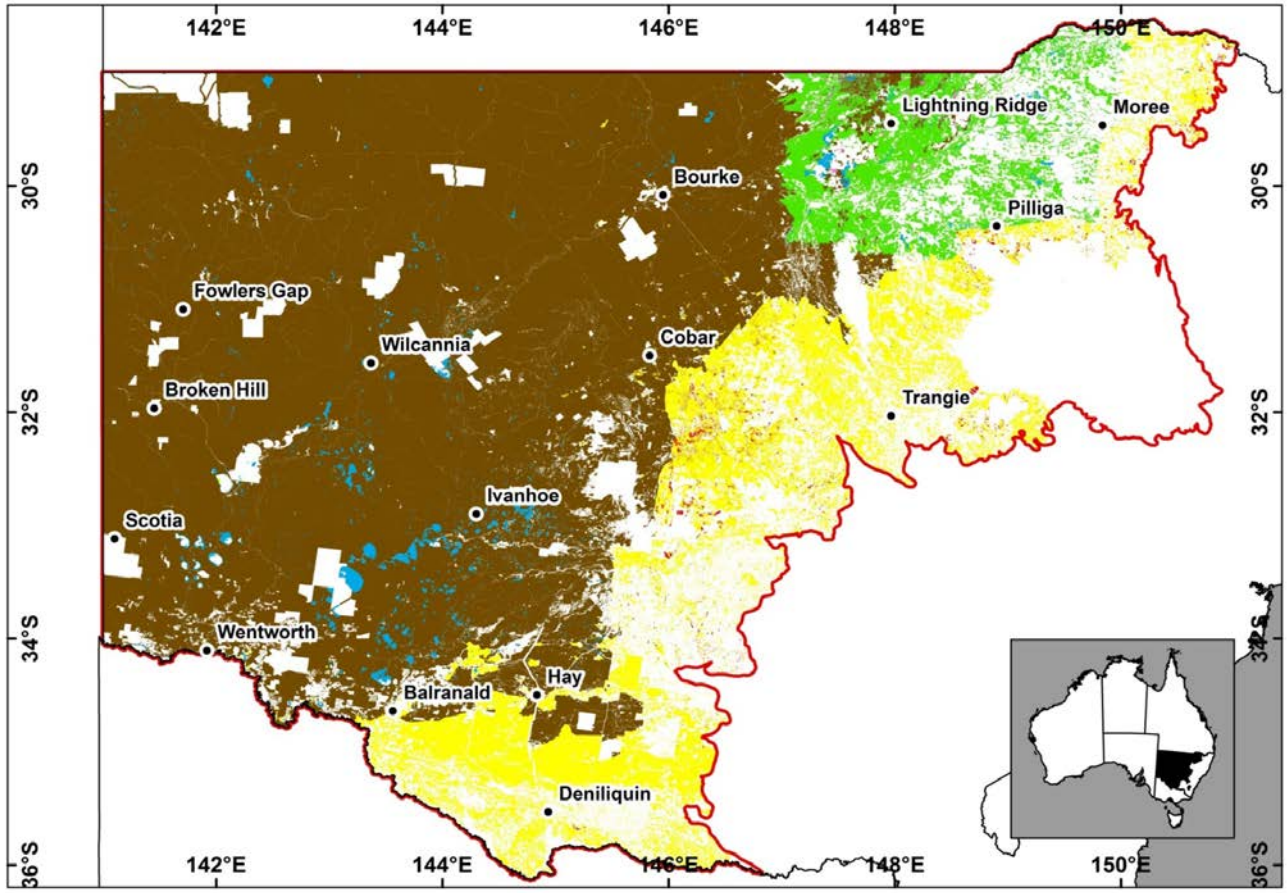


Figure 10-3. Distribution of main land-use types. Yellow= 'volunteer' pasture with no woody cover (not included in study). Brown= 'rangeland'; light green= 'unmodified native vegetation'; light blue= 'riparian and wetland'; pink= 'recently cleared'; red= 'volunteer, naturalised or improved, with >30% woody regeneration'; dark blue= 'secondary grassland in forest'; dark green= woodland, unmodified'.

LANDSAT imagery with 25 m pixels, were from NSW Office of Environment and Heritage (Table A-V-1). The recently deforested land (from 1989–2007) was selected for viewing correlations between biomass and environmental variables because it avoided corruption of the correlation by interstitial grasslands. Data for deforestation prior to 1989 did not include dates and were grouped with other forms of forest degradation. Representative vegetation contrasts observed in remotely-sensed imagery were qualitatively examined on-ground in a subset of areas State-wide, between 1996 and 2006. The locations included long-term grazing exclosures, a carbon sampling experiment, and numerous roadside contrasts observed during journeys for such work.

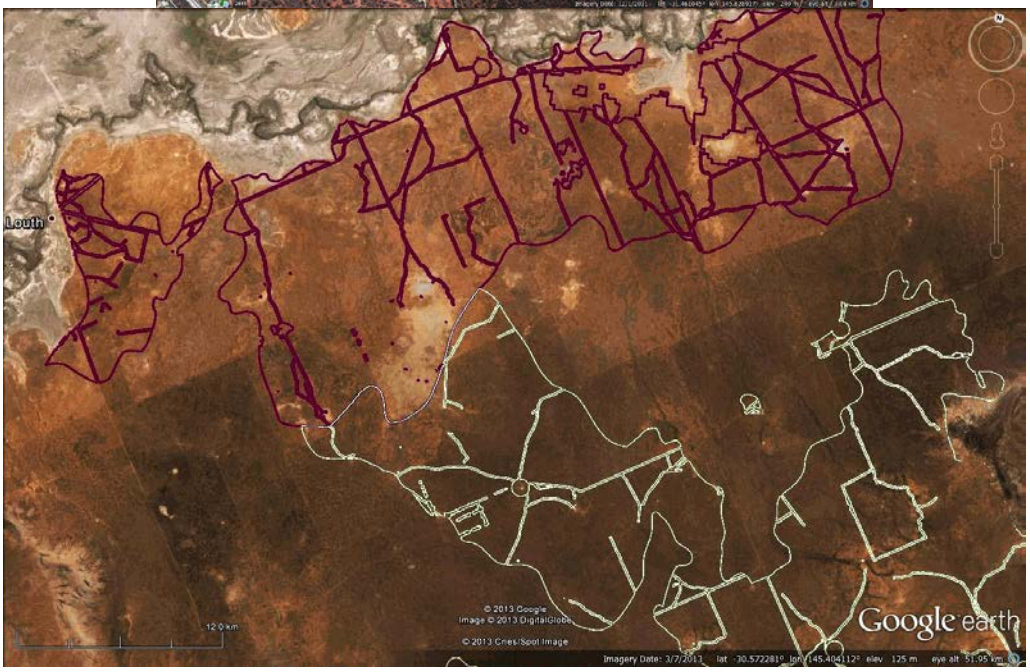
Representative areas that theoretically had minimal attrition of vegetation— the benchmarks— were selected from 'remnant' ecosystems by a rigorous process of detecting and eliminating land that had been anthropogenically denuded of woody vegetation (Figure 10-4, Figures A-V-1–3). Infrastructure, urbanisation, and roads were subtracted, all with buffer zones due to the possibility of pixel misplacement and because neighbouring vegetation was usually depleted. Buffer distances were: deforestation 50 m; minor roads 20 m; major roads 50 m; homesteads and towns 500 m. There were 226 ecosystem types with remnant vegetation, covering 28 Mha of the greater rangeland zone in NSW.

(a)

(b)


Google earth

(c)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© 2013 Google
Image © 2013 DigitalGlobe
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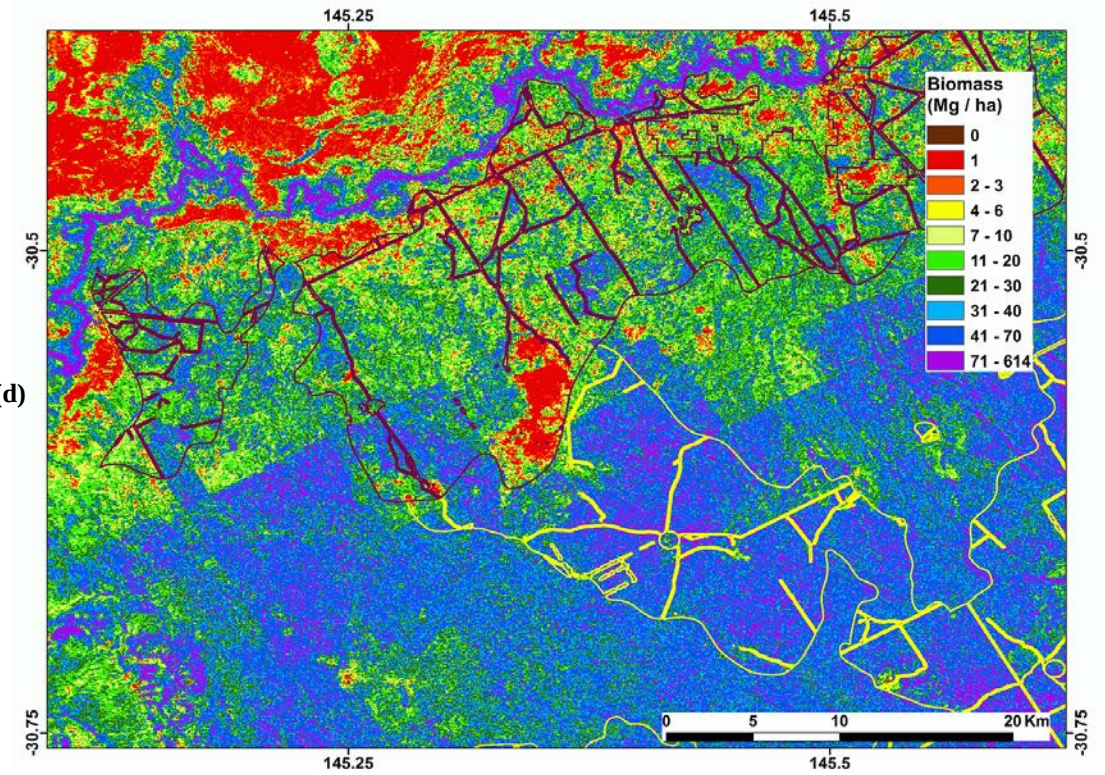
Google earth

(d)



(e)

Figure 10-4. Example degradation of remnant ecosystems (biomass benchmarks). **(a)** Location: ~86 km NW of Bourke, Lat/Long -29.523°/145.406°. GoogleEarth© image of 23-Sep-2006, showing deforestation, livestock tracks, vehicular tracks, grazing land (deforested and non-deforested), roads and farm infrastructure, and typical forest attrition within the ecosystem remnant. **(b)** Finer level

attrition of remnant ecosystem, centre: -31.460848°/145.829238°. **(c)** Centre: [-30.554402 S, 145.472336 E]. Nearly the whole ecosystem 'Mid-Darling Plains' (coloured burgundy), the closest to the river, has suffered substantial attrition. Parts of the neighbouring ecosystem 'Cobar Downs' (yellow outlines) were therefore incorrectly assigned higher potential biomass than the 'Mid-Darling Plains' to the north. **(d)** The ALOS data show that the fenceline contrasts are due to differences in woody biomass, which is confirmed in the GoogleEarth© closeup. **(e)** Centre: [-30.622615, 145.186126], closeup of left of (c). The fenceline contrast is due to a loss of mature trees to the NW of the fence-line. Centre of the GoogleEarth© finer-detail image shows the loss has been replaced by a multitude of smaller trees or bushes (i.e. 'woody thickening').

Data on potential biomass were available in two forms: a (~5 km pixel) NOAA-AVHRR-derived layer from Berry and Roderick (2006); and a higher resolution layer (0.0025×0.0025°, i.e.~250×250 m pixels) from the Commonwealth Department of Climate Change and Energy Efficiency, as used in the National Carbon Accounting System (NCAS) (Richards and Brack, 2004; Waterworth and Richards, 2008). Data on extant aboveground biomass were also available in two forms: a (~5 km pixel) NOAA-AVHRR-derived layer from Berry and Roderick (2006); and ALOS PALSAR radar data (acquired in 2009) pre-processed to yield extant, aboveground, woody biomass (50 x 50m pixels) (Armston et al., 2010; Lucas et al., 2010) (from NSW Office of Environment and Heritage). The results from NOAA-AVHRR and PALSAR data were compared.

The ALOS (extant) and NCAS (potential) biomass values were compared for remnant ecosystems, i.e. where they should have been equal, and aberrations were interrogated. ALOS data are potentially more reliable than NCAS data as they rely less on growth modelling, but exceptions to this are discussed in the appropriate sections. Areas of obvious incongruence, e.g. where canopy loss was >50% or infrastructure was present, were eliminated. The same buffer distances as described above for benchmark detection were applied. (Figure 10-4, Figures A-V-1–3) Much of the fence-line clearing for 'infrastructure' was ~60 m wide (Figure 10-4.a). Only about half such fence-line clearing was delineated as deforested in the LANDSAT-

Figure 10-5. Example areas likely to have undergone substantial deforestation at least two decades ago; minimal recruitment of new saplings. The native seedbank has mostly likely been depleted. Note: (i) biomass stocks are not replenished, (ii) roots of deforested individuals are not replenished and slowly decompose— causing long-term, net carbon efflux, and (iii) soil carbon stocks slowly decline without continual input from further tree decomposition products. Reforestation would constitute a prime re-sequestration opportunity. **(a)** Cobar peneplain, NSW. There is a green flush (annuals rather than native perennials) after recent rain, following overgrazing, which can give a "false positive" indication in some two-dimensional remotely sensed imagery such as LANDSAT, although not in LiDAR or Radar. **(b)** Ivanhoe area, NSW.

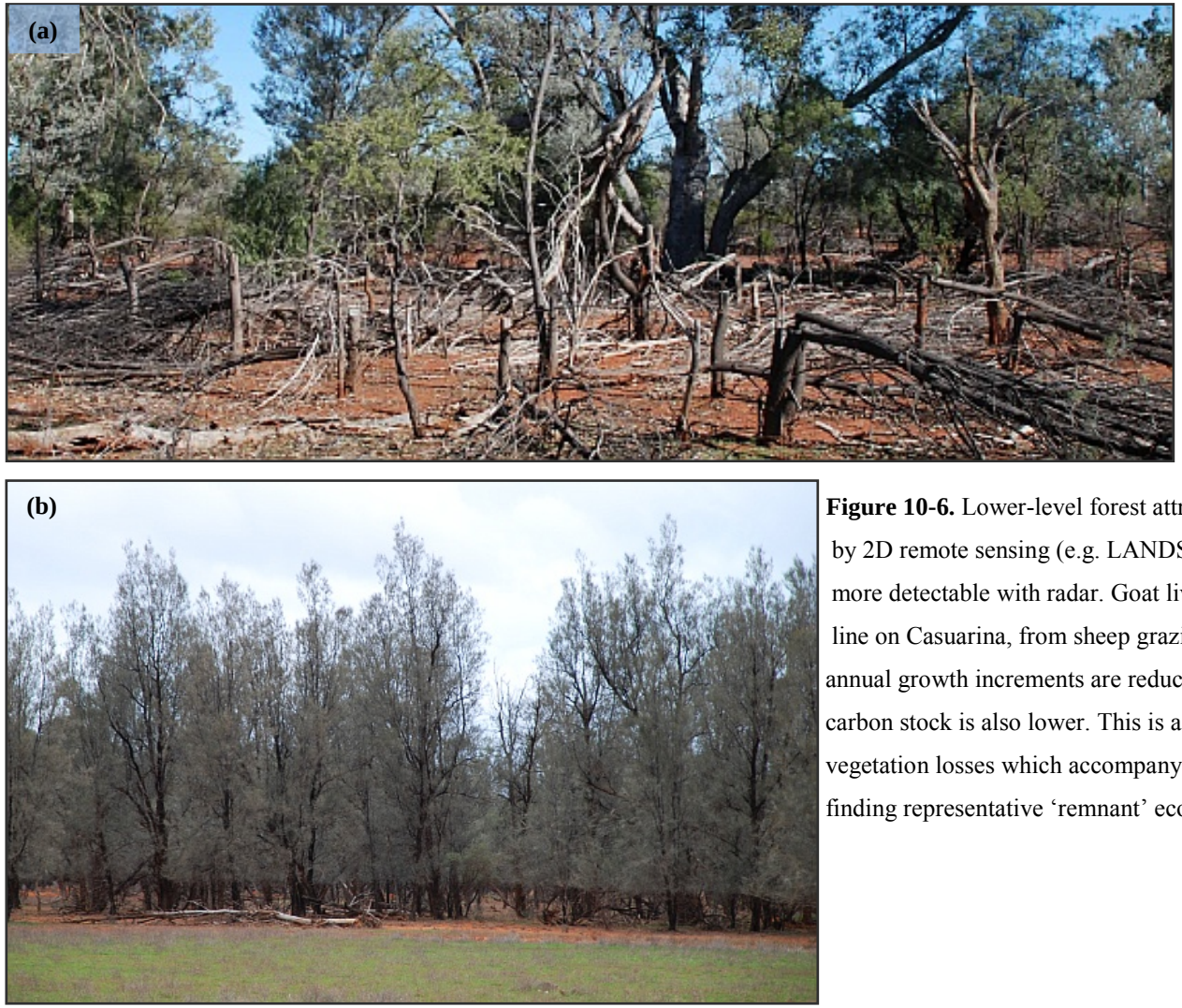


Figure 10-6. Lower-level forest attrition. **(a)** Selective thinning, difficult to detect by 2D remote sensing (e.g. LANDSAT) if there is increased grass growth, but more detectable with radar. Goat livestock. Cobar Peneplain, NSW. **(b)** Browse line on Casuarina, from sheep grazing. Murray-Darling depression, NSW. The annual growth increments are reduced by such grazing, hence the long-term carbon stock is also lower. This is a minor but contributing factor to the observed vegetation losses which accompanying forest degradation and the problem of finding representative 'remnant' ecosystems.

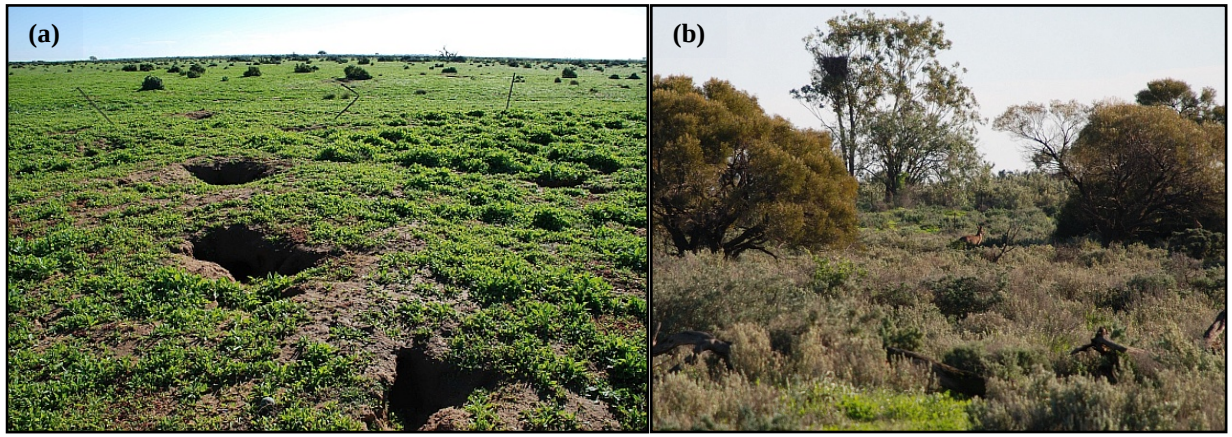


Figure 10-7. Stock travelling route ('long paddock'), Hay plains, NSW. **(a)** Deforestation with loss of saltbush and trees, replacement by weeds and rabbits—reducing carbon stocks. **(b)** Neighbouring area with livestock excluded.

Figure 10-8. Loss of topsoil (bared roots, soil erosion and net SOC depletion) in an overgrazed sheep paddock, due to either overgrazing or possibly livestock camping (for shade or shelter). (Note green flush of annual grass (indicator of overgrazing) following recent rains.) Ivanhoe area, NSW.

derived data. Ground-truthing showed that it was used for livestock grazing or mustering. It was delineated as rangeland grazing in the land-use data layer.

Many of the benchmark areas had reduced woody biomass, with numerous minor [unlisted] roads, vehicular tracks, intentional thinning, overgrazing and other forms of attrition (Figure 10-4, Figures A-V-1–3). These complications were too pervasive to subtract from representative remnant ecosystems and it was recognised they would affect calculation of potential biomass. The degree of biomass attrition varied between benchmarks. Benchmarks near major rivers were particularly depleted, as confirmed from ALOS data and GoogleEarth© imagery of fence-line contrasts (e.g. Figure 10-4.c, d). Some were so uniformly and strongly below their potential, that their extant biomass (from ALOS data) appeared lower than their non-riparian neighbours, and thus caused the highest error margins in the calculations. Some highly degraded areas had some loss replaced by a large number of smaller trees or shrubs, i.e. 'woody thickening' (Figure 10-4.c, d), which is often linked to overgrazing during droughts that are followed by heavy rains (Allen, 1983; McKeon and Hall, 2001; Laliberte, 2004; Svejcar et al., 2008). Variability within benchmarks added to error margins.

Ongoing annual emissions for the NSW rangeland were estimated from the annual emissions calculated in Chapter 9 for Australian commercial rangelands, apportioned on an area-basis for NSW. Firstly though, the deforestation in QLD (20.47 Tg $year^{-1}$) and savannah burning (as deployed in tropical Australia) were subtracted. The estimate of Chapter 9 did not include the NSW deforestation effects calculated here, which were added in. Also the efflux with general degradation of national Mulga Lands (Chapter 9) was apportioned to the 6.319 Mha of Mulga Lands in the NSW commercial rangelands. The difference in potential concentration of biomass between the QLD and the NSW Mulga Lands was calculated from the NCAS layer (QLD:NSW ratio of 1.29:1) and applied to the apportioning of Mulga Lands degradation for NSW.

SOC emission with deforestation for NSW were estimated on a pro-rata area basis from the QLD sums and the ratio of QLD:NSW SOC concentration (1.15) in Chapter 9, and compared to the -17% of SOC emitted over 30 years (Chapter 8) for QLD deforestation. That figure was applied to typical NSW deforestation rates over 50 years to get an annual SOC efflux.

The long-term effect of continued anthropogenic soil erosion and the benefit of land rehabilitation was modelled and graphed for Australian rangeland in Chapter 9 and those values can be approximated pro-rata, for NSW. Land rehabilitation does not immediately stop soil erosion. If rehabilitation (most likely requiring financial input) was started in 2014, then the total SOC emission (to 0.3 m depth) due to commercial grazing to date would be 110 (±40) Tg, peaking in 2400, including 99 (±40) Tg after 2014. The true error margins for these estimates are high, possibly around 90% (an estimate based on variability in the effects of management and climate change on ecology, for example, erosion effects as discussed in section 9.5.1). The SOC emissions with rehabilitation beginning in 2014 and all anthropogenic erosion curtailed by 2053 were calculated and compared with emissions with business as usual (i.e. with current erosions rates).

10.3.3 Biomass growth modelling

Growth modelling of rangeland recovery suffers from insufficient data (Dwyer et al., 2009). Linear growth is often assumed, despite acknowledged variation with age, and only a single rate of sequestration is typically reported (e.g. Bray and Golden, 2008; Dwyer et al., 2009; Fensham and Guymer, 2009; Dean et al., 2012c). The non-linear growth formulation typically used in even-aged forestry (e.g. Waterworth et al., 2007) has been used to model reforestation by semiarid plantations (Paterson and Bryan, 2012). The function has two dependent variables: the potential biomass and the age of fastest growth, called ‘G’— the inflection point of the growth curve. It could be used to model thickening and rangeland reforestation (slow, uneven-aged growth), with adjustment of G:

$$currentC = potentialC \times \exp\left(-\left(2G - 1.25\right)/t\right) \qquad \text{Eq10-1}$$

where *currentC* is the C (in Mg ha^{-1}) at time *t* (in years), *potentialC* is the maximum C attainable at the site (the benchmark in Mg ha^{-1}), and *G* is the inflection point of the curve— the age of maximum current annual increment for the stand (CAI) (in years). A range of values of *G* were tested here (Figure 10-9).

Fitting Eq10-1 to the stand-level thickening work of Hibbard et al. (2003) and Chew and Chew (1965) gave values for G of 29 and 89 years, respectively, even though G for individual trees in the latter study was 11 years (i.e. closer to that for plantations). The data of Kroeger *et al*. (2005) gave a value of G near 80 years for reforestation of oak savannah in Californian Mediterranean rangeland. Notably though, the function designed for high stocking rates and even-aged germination (i.e. Eq10-1), did not fit the reality of thickening well. A better fit for stand growth with thickening (slower start but approaching the asymptote near the same time) was found using the more-flexible logistic function from Chapter 8 and that is applied here, in addition to Eq10-1, to determine sequestration with reforestation:

$$currentC = potentialC \times \left(1 - \frac{1}{1 + \left(t/b\right)^{c}}\right) \qquad \text{Eq10.2}$$

where *currentC* is the C (in Mg ha^{-1}) at time *t* (in years), *potentialC* is the maximum C attainable at the site (the benchmark in Mg ha^{-1}), *b* is the time (in years) when half the potential is reached, and *c* is a dimensionless parameter that determines the steepness of the curve around the inflection point. Parameter values that fit the empirical and modelling work of Hibbard et al. (2003) and Chew and Chew (1965) (in Texas and Arizona respectively) on woody thickening are *b*=75.2799 years, and *c*=2.8389— those values were used herein (Figure 10-9).

In both Eq10-1 and Eq10-2 the biomass [as a function of age] is linearly proportional to the potential biomass. Consequently the spatial average flux, across different potentials, is linearly proportional to the spatial-average potential. That

simplified the calculations, with all sequestration rate curves, for differently sized areas, being linearly proportional. For calculation of re-sequestration the potential biomass of benchmarks was deduced from ALOS data only, except for areas recently deforested, where the average of ALOS and NCAS was used, as there the extant remnant vegetation was considered less representative.

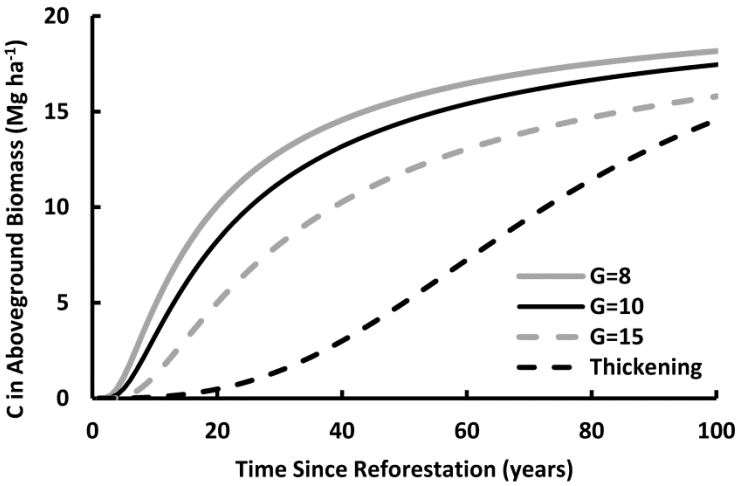


Figure 10-9. Possible growth curves for the recently deforested 80,667 ha of NSW rangeland, showing effect of variation in year of maximum growth (G) for Eq10-1, and use of the lower rate 'Thickening' (Eq10-2) modelled on uneven-aged stand development via woody-thickening. The three higher rates are based on growth curves for plantations in southern Australia (although scaled to local potential biomass).

10.3.4 Livestock modelling

In order to gauge opportunities from livestock stocking-level adjustment, spatial data on livestock carrying capacity and extant stocking levels were sought. The most recent data, supplied by David Burcell (NSW Livestock Health and Practices Authority (LHPA), personal communication, 2011) pertained to 21.2 Mha of the 30.57 Mha of NSW commercial rangeland, on one voluntary census night in 2010. Relationships were examined between notional (i.e. advised carrying capacities) and actual stocking rates and between these and environmental variables. Those

relationships were spatially interpolated to provide stocking levels for areas without stocking data. The census data may have been subjective to some degree, and included such aberrations as stock from one large and frequently used paddock mustered into a smaller and less-frequently used paddock (or even shipped elsewhere at the time), which would for example weaken the correlation between biomass of a large paddock and its average stocking level.

NSW rangelands are currently mostly stocked with sheep, goats and cattle. Livestock census data were converted into 'dry sheep equivalents' (DSE) per hectare (with 1 DSE being one 50 kg wether or non-lactating ewe).

Data on farm [net] profits were obtained from Salahadin Khairo (Department of Primary Industries NSW, personal communication; 2011) and Khairo et al. (2008). These data were from three typical properties in different parts of the NSW rangelands. They included all typical running costs and incomes, apart from inputs from government grants, subsidies and allowances. Data for the three typical farms were averaged. The mean profit on an area basis was \$3.8 (±1.8) ha^{-1} $year^{-1}$ (standard deviation of mean in brackets), the mean profit per livestock unit was \$11.5 (±2.6) DSE^{-1} $year^{-1}$ and the average stocking rate was 0.33 (±0.17) DSE ha^{-1}.

The methane emissions from all types of livestock were converted to mass per DSE per unit time. A conservative estimate of the methane emission per DSE was 11.1 g DSE^{-1} day^{-1}, based on literature values (Leuning et al., 1999). With methane having a global warming potential of 23 times that of CO_2 over a 100 year time frame (reports citing 21 to 25 (IPCC, 2007; UNFCC, 2014)), the presence of one DSE $year^{-1}$ equates to a C efflux of 0.025449 Mg DSE^{-1} $year^{-1}$. That conversion factor was used in estimates of avoided emissions upon destocking. A carbon price of AUD\$10 Mg^{-1} of CO_2-e equates to a methane emission cost of \$0.93 DSE^{-1} $year^{-1}$.

The comparisons between carbon and grazing profits were not adjusted for discount rates as the sequestration mechanism requires minimal investment and the timing of potential income or fines for carbon flux is a matter for policy construction. Also, species that reforest without planting (passive reforestation) (woody-thickening

growth rates were used) are more likely to be acclimatised to the local fire regime and will not be as easily killed by fire as are many plantations, thereby lowering the need for one of the discounts typically applied to plantation schemes.

10.4 Results

10.4.1 Climate and livestock

The main biomes in NSW are temperate (savannahs, grasslands, or shrublands) with lesser amounts of Mediterranean (forest, woodland and scrub) and xeric biomes (Figure A-V-16). Approximately 25% of all Australian commercial rangeland has similar biomes and climate to those of NSW (Table A-V-8, Figure A-V-16,17). The most widespread biome in NSW is the temperate grasslands, savannahs or shrublands. Significant areas of that biome are also found in QLD (Australia) and Texas (USA) (Figure A-V-16,17.b). The areas of xeric and Mediterranean biomes in NSW are approximately equal. The ecosystem distribution data revealed that only ~1 (±0.3)% of NSW commercial rangelands are natural grassland and that even those have scattered trees or groves of trees or saltbush, which concurs with the historical qualitative analysis (Allen, 1983). In terms of absolute areas, after NSW the next biggest area of similar Mediterranean biome is in northern Africa, bordering the Mediterranean (Figure A-V-17.a). Mexico and southern Africa have the most area of similar xeric biome. Countries with significant portions of their rangeland similar to NSW are: Syria with temperate shrublands, grasslands or savannahs; Malta with Mediterranean forests, woodlands, or scrub; and Botswana with desert, or xeric shrublands (Table A-V-8, 9, Figure A-V-17).

For land in the study area deforested from 1989–2007, potential biomass and rainfall were closely related both increasing from west to east, and SOC and temperature were closely related, both increasing from north to south (Figure 10-11, Figure A-V-5.a,b). As the time since deforestation was short compared with SOC half-lives, these

correlations would have resulted from the woody vegetation state (its decomposition of necromass over millennia) rather than the post-deforestation, grassland state.

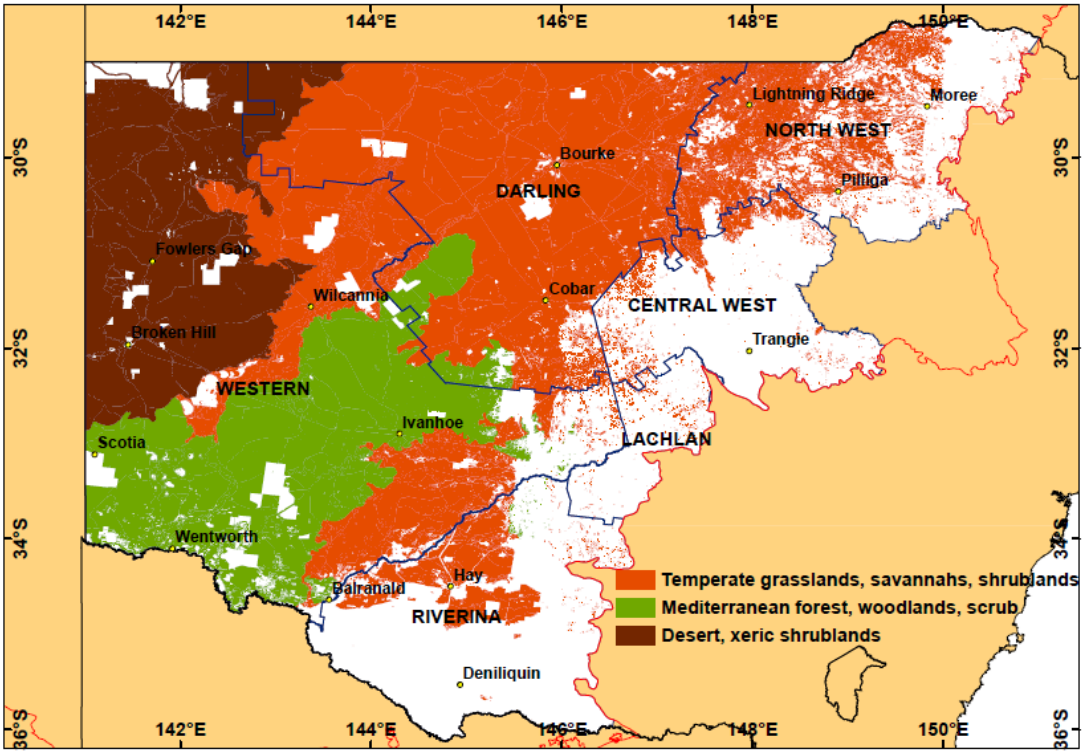


Figure 10-10. NSW rangeland, biomes (coloured). Biome GIS data from Olson et al. (2001). Geographic projection.

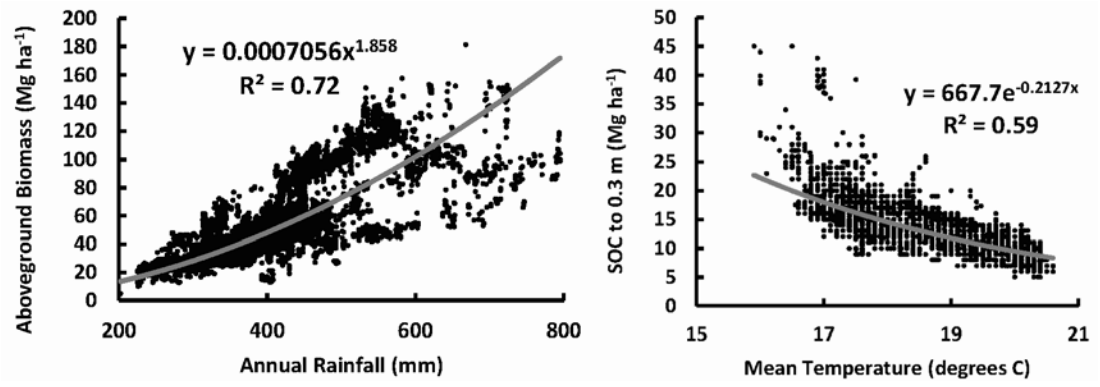


Figure 10-11. Relationships between potential biomass, SOC and environmental variables, for land deforested from 1989–2007.

Average annual rainfall, temperature, aboveground dry biomass, and SOC (to 0.3 m) (all area-weighted) for the NSW commercial rangeland were 320 mm year^{-1}, 19° C, 36 Mg ha^{-1}, and 10.3 Mg ha^{-1} respectively. Approximately half the area is arid (47%), the remainder semiarid (Figure 10-10).

There was a positive relationship between notional carrying capacity and rainfall (Figure A-V-7). A lower correlation was found between actual stocking levels and rainfall. The notional carrying capacity formula found was:

$$NCC = 5.992 \times 10^{-8} rain^{2.6808} \qquad r^2 = 0.66 \qquad \text{Eq10-3}$$

where *NCC* is in units of DSE ha^{-1} (dry sheep equivalent per hectare) and *rain* is the mean annual rainfall in mm. And the actual livestock stocking rate formula found was:

$$DSEa = 6.338 \times 10^{-8} rain^{2.6005} \qquad r^2 = 0.31 \qquad \text{Eq10-4}$$

where *DSEa* is in units of DSE ha^{-1} (dry sheep equivalent per hectare) and *rain* is the annual rainfall in mm.

On average the actual stocking rates were 30% below notional capacity. Note however that only approximately two thirds of livestock managers reported their actual livestock numbers, so the relationship could differ when including the whole State if those who did not report stocking had a particular management style.

The total number of DSE over the 30.57 Mha of NSW commercial rangelands was 7.1755 million if the land is stocked at the actual stocking level, or 10.895 million if the land is stocked at the notional-level (averaging 9 (±2) million). That corresponds to 0.2347 and 0.3564 DSE ha^{-1}, respectively for the actual and notional stocking levels per hectare, averaging 0.29 (±0.06) DSE ha^{-1}, which is close to the average of 0.3307 DSE ha^{-1} from the three regional examples in Khairo et al. (2008). The net profit from the livestock NSW-wide, assuming an average farm, is given by: 9.0

(±1.2) million DSE multiplied by $11.5 (±2.6) DSE^{-1} $year^{-1}$ equalling $100 (±30)M $year^{-1}$.

10.4.2 Biomass change

Over all land uses in NSW, the change in aboveground biomass, as determined using the ALOS data and remnant ecosystems, was -867.98 Tg, i.e. 434 Tg of C emitted. For commercial rangeland within that area the change was -51.695 Tg (i.e. -26 (±9) Tg of C (Figure 10-12, Figure 10-13, Figure 10-14.a). The nominal 30% error margin is based on spatial variability within ecosystems. Various uncertainties were included in this error estimate, such as the representativeness of remnant ecosystems, and from the comparison of ALOS and NCAS values, e.g. Figure A-V-12 in Appendix V. Using a root:shoot ratio of 1:3 (25%) the net emitted C in total biomass was then 35 (±10) Tg. When using the NOAA-AVHRR data in place of the ALOS data the net

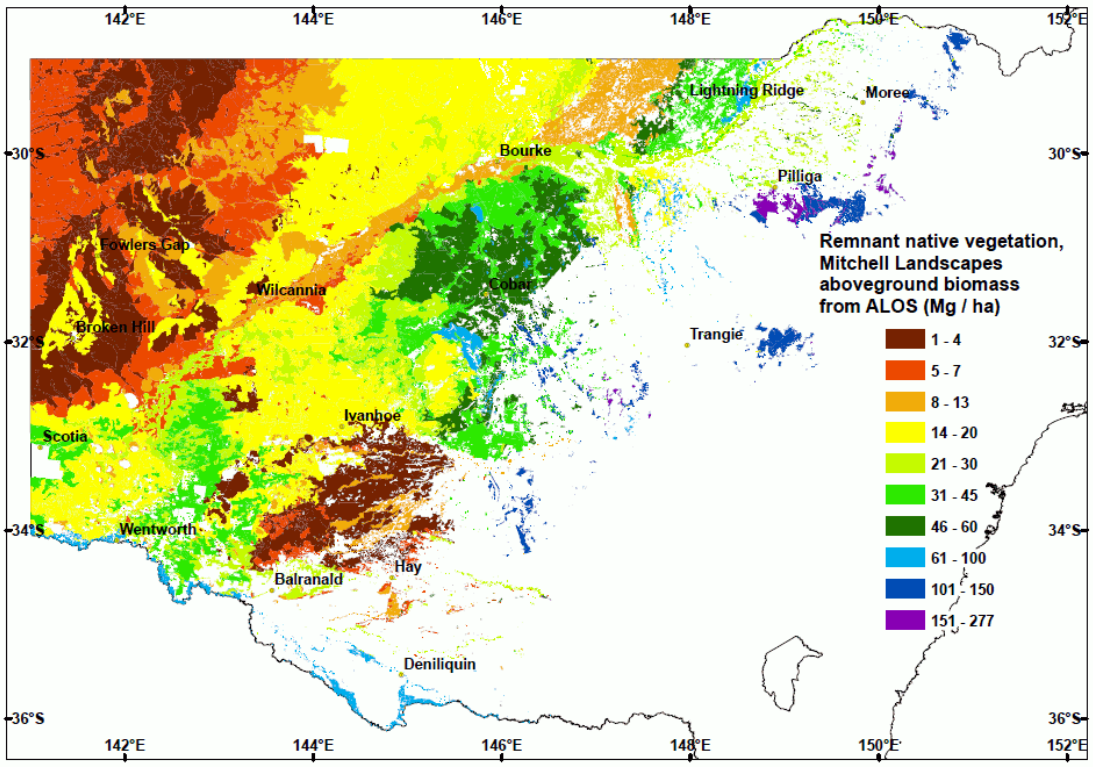


Figure 10-12. Representative aboveground biomass from ALOS data, for the remnant ecosystems, grouped by ecosystem-type (Mitchell Landscape).

emitted C in total biomass was 35.50 Tg. These are similar results given that the error margin of current biomass stocks from Berry and Roderick (2006) is in the order of 15% nationwide but up to 50% in some locations (e.g. near seasonal salt lakes).

For remnant ecosystems, the ALOS biomass values were mostly lower than the NCAS values, on average by 14% (Figure A-V-12).

If instead of the net change in biomass from the ALOS analysis, only land with biomass below that of the remnant ecosystems is assessed, namely 17.90 Mha, the emitted C from aboveground biomass is 61 (±18) Tg (Table 10-1, Figure 10-14.a, Table A-V-3). That value constitutes an amount that could be sequestered, by bringing that 17.901 Mha up to the remnant values for each ecosystem type. It is likely that somewhere between the net and the deficit alone could be sequestered in aboveground biomass, i.e. 26–61 Tg of C. If root biomass is included that range is 34–81 Tg, i.e. 58 (±24) Tg— similar to the value of 41 Tg derived from lower resolution data by Dean *et al*. (2009).

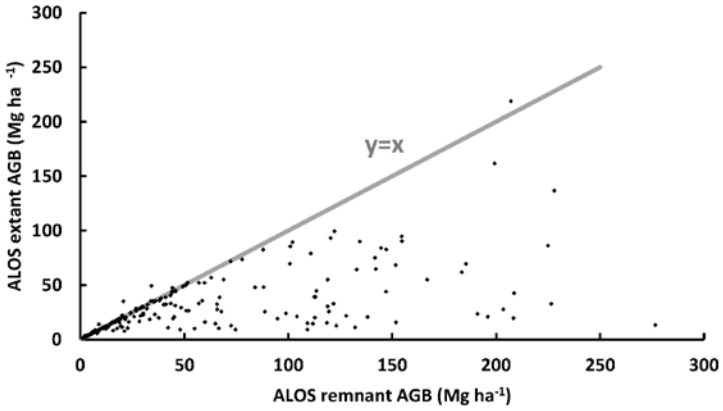


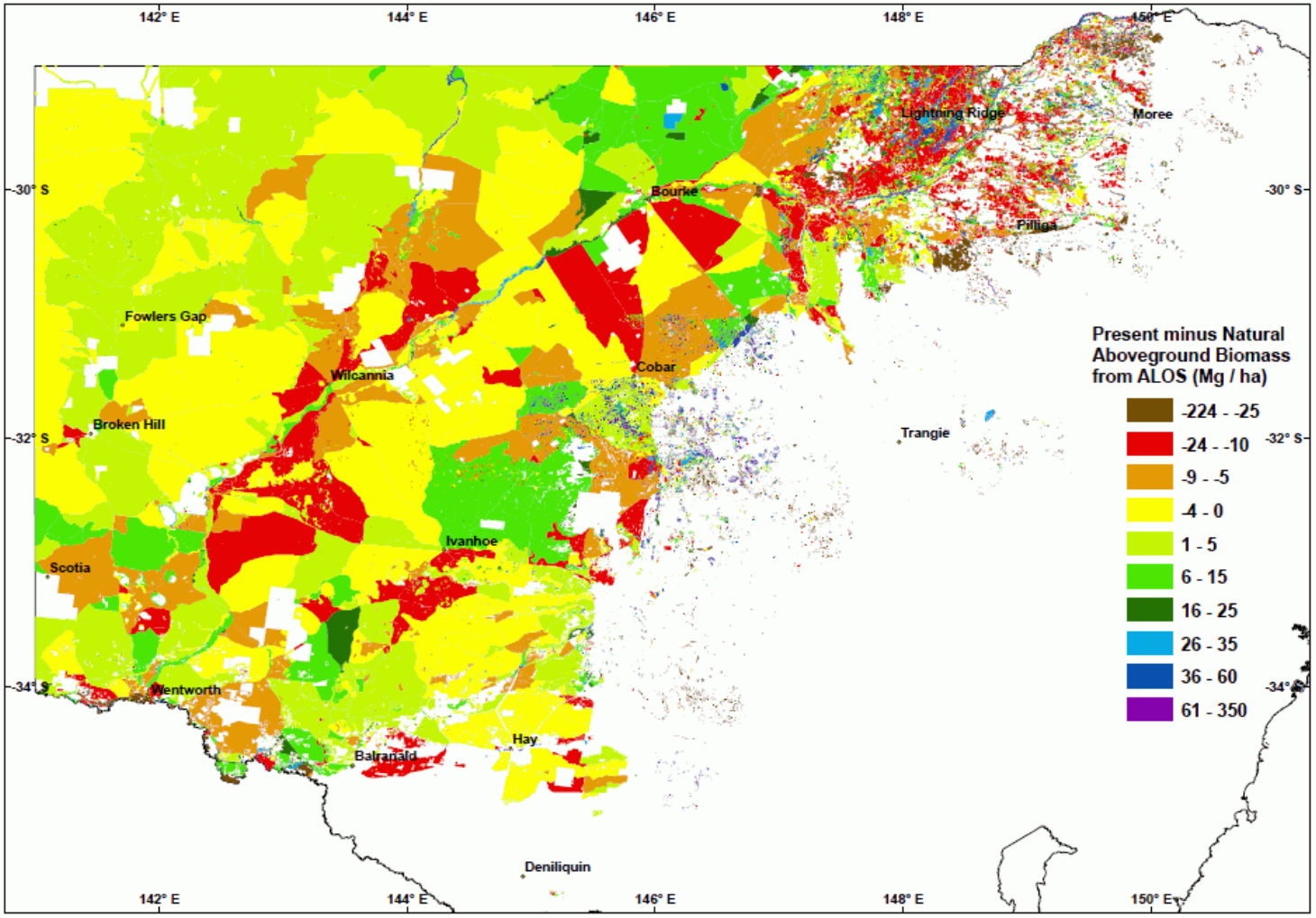
Figure 10-13. Aboveground biomass (from ALOS) for the ecosystems: average-extant (present) versus remnant. As expected due to forest attrition with commercial grazing, the average-extant biomasses are mostly below the remnant values. A few ecosystems presently have higher biomass on average than in their remnants, mostly due to the purely 'riparian' land-use category not being classed as 'remnant' within the 'rangeland' land-use category (confirmed in GoogleEarth imagery).

To allow revegetation, an estimated 4.49-to-6.82 MDSE would need to be removed (at actual and notional stocking rates respectively). This corresponds to a reduction in profit of \$65 (±34)M year^{-1} at the average stocking rate (using the average farm profit of \$11.5 (±2.6) DSE^{-1} year^{-1}). Conversely, the income from re-sequestering the 58 (±24) Tg of aboveground C is \$2100 (±900)M (\$2800 (±1000)M including roots). This income is distributed over several decades, the timelines of which are discussed below.

Table 10-1. Levels of rangeland forest degradation targeted, and potential C re-sequestration. The C values listed are spatial averages: higher deficit (re-sequestration) is likely in the more denuded patches.

Commercial rangeland type	Area (Mha)	C deficit in aboveground biomass (Tg)	Deficit per hectare (Mg ha^{-1})
NSW, State-wide	30.57	26 (±9)	0.85 (±0.29)
Land below remnant value	17.90	61 (±18)	3.4 (±1.0)
Land below extant average	12.85	37 (±11)	2.9 (±0.9)
Land below remnant and extant average	2.219	3.7 (±1.1)	1.7 (±0.5)
Recent deforestation	0.08450	1.8 (±0.2)	21 (±6)

An alternative re-sequestration goal could be partial rehabilitation/replenishment, enough to bring the poorest vegetation condition back up to the local, extant average for each ecosystem type. The carbon in aboveground biomass missing from the 12.85 Mha that are below the local, extant-averages (Figure 10-14.b), is 37 (±11) Tg (Table 10-1, Table A-V-4), or 50 (±15) Tg if C including roots (equating to AUD\$190 (±60)M). To revegetate this area a total of 3.14–4.77 MDSE would need to be destocked (for actual and notional stocking rates respectively), equating to -\$46 (±12) M.yr^{-1} in livestock profits. The avoided C effluxes from that livestock's methane are 1 (±0.3) Tg over ten years, and 10 (±3) Tg over 100 years. The latter value is comparable with the magnitude of the error margin in potential sequestration


142° E
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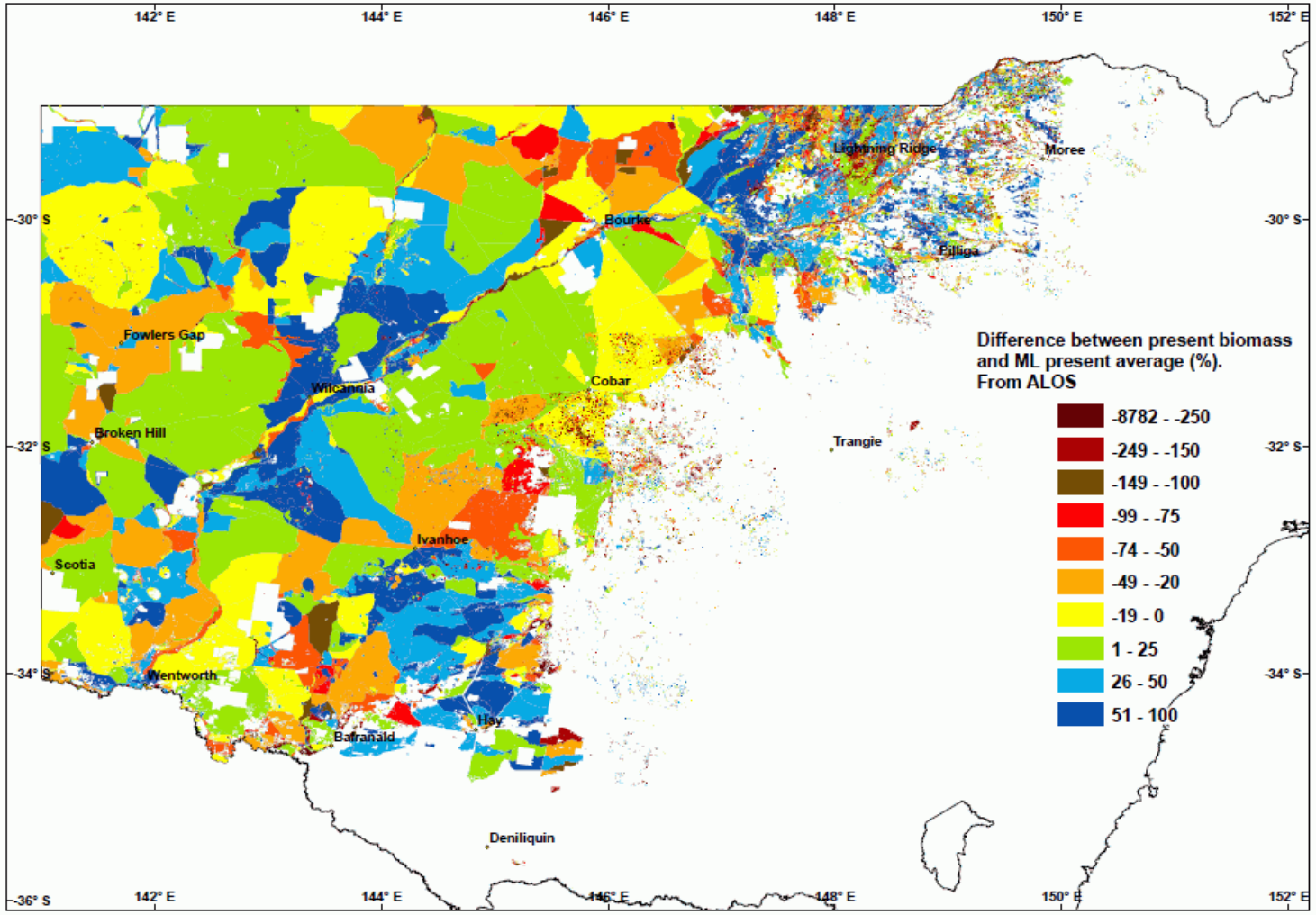
32° S
34° S
Lightning Ridge
Moree
Bourke
Pilliga
Fowlers Gap
Wilcannia
Cobar
Broken Hill
Trangie
Ivanhoe
Scotia
Wentworth
Hay
Balranald
Deniliquin
Present minus Natural Aboveground Biomass from ALOS (Mg / ha)
-224 - -25
-24 - -10
-9 - -5
-4 - 0
1 - 5
6 - 15
16 - 25
26 - 35
36 - 60
61 - 350

(a)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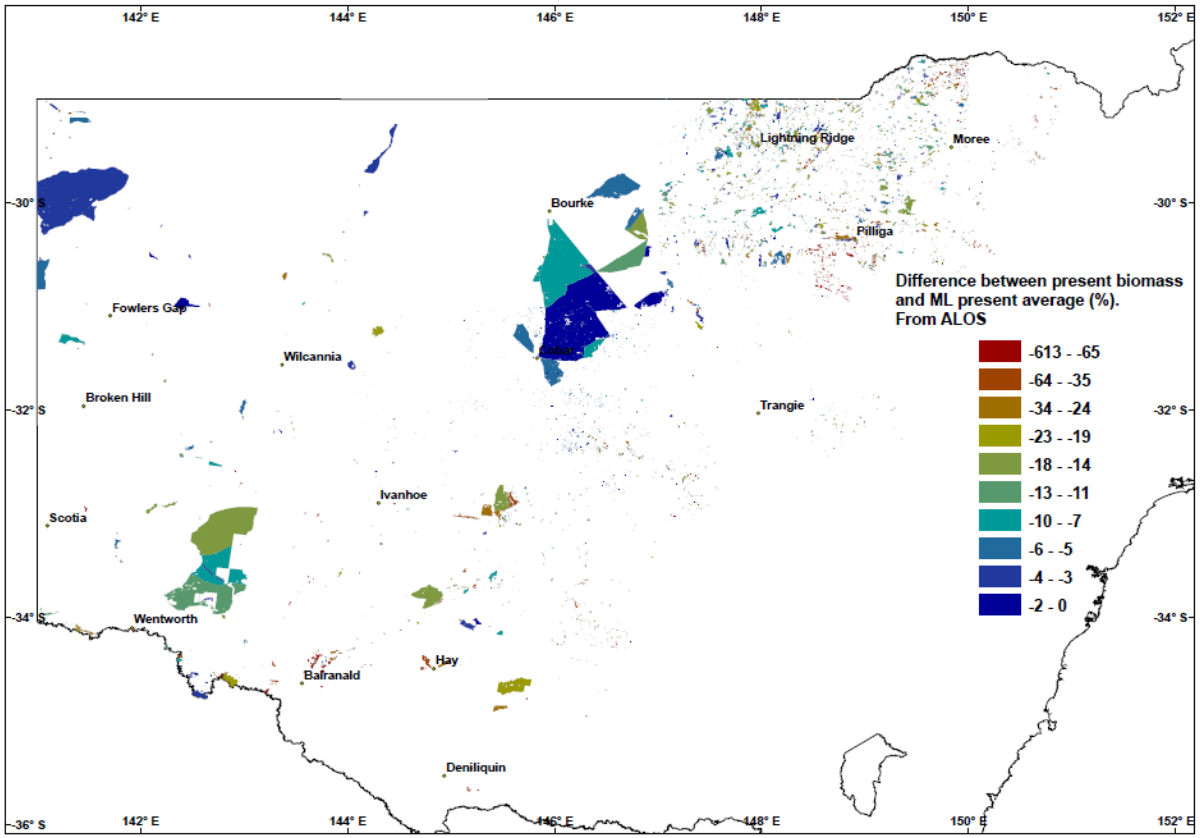
Pilliga
Fowlers Gap
Wilcannia
Cobar
Broken Hill
Trangie
Ivanhoe
Scotia
Wentworth
Hay
Balranald
Deniliquin
Difference between present biomass and ML present average (%). From ALOS
-8782 - -250
-249 - -150
-149 - -100
-99 - -75
-74 - -50
-49 - -20
-19 - 0
1 - 25
26 - 50
51 - 100

(b)



(c)

Figure 10-14. Regions of biomass loss. Negative values indicate a deficit. **(a)** Extant minus remnant. The main negative area is a SW-NE band following the Darling River. **(b)** Extant minus average-extant, for each ecosystem-type. Compared with (a), this is a local, relative gauge. **(c)** Negative areas in both (a) and (b). The most consistently negative region is between Cobar and Bourke (upper centre).

in biomass (Table 10-1). Thus, C re-sequestration does not recover forgone livestock income.

A total of 2.219 Mha of commercial rangeland (Figure 10-14.c) scored negatively when compared with both remnant vegetation (within each ecosystem type) and extant averages (for ecosystem types). Its carbon deficit of aboveground biomass was 3.7 (±1.1) Tg, or 5.0 (±1.5) Tg including roots. This represents the absolute minimum of C that could be re-sequestered. Several large regions consistently showed as negative (Figure 10-14), across all three NSW-wide analyses, the largest being 0.68 Mha, east of the Cobar–Bourke road.

For the recent deforestation (1989–2007, 84,479 ha), the emitted C from aboveground biomass was 1.59-to-1.96 Tg (from ALOS and NCAS potential biomass respectively), (Table 10-1, Table 10-2, Table A-V-5). That represents what could be sequestered upon reforestation: an average of 1.8 (±0.2) Tg (2.3 (±0.3) Tg with roots), or 21 (±2) Mg ha^{-1}. A plausible error margin is likely to be at least twice the difference between the two estimates. If the emitted carbon as a result of land management activities is brought to account, the bill is $87 (±26)M (including roots), or $1000 (±300) ha^{-1}. If using farm [net] profits from livestock grazing, it would take 270 (±80) years to repay, either by the NSW government or the pastoralist on leasehold land, or by the owners of freehold land. (The area is 11% freehold, 89% leasehold.)

Table 10-2. Summary statistics for commercial rangeland deforested from 1989–2007.

Tenure	Area deforested (ha)	C density in aboveground biomass (Mg ha^{-1}) (average of NCAS and ALOS)	Livestock density DSE ha^{-1}	Cost of total C emitted (AUD$ million)
Freehold	9,547	38 (±11)	0.792	18 (±5)
Leasehold	74,948	19 (±6)	0.305	69 (±21)
Total/average	84,495	21 (±6)	0.360	87 (±20)

10.4.3 Re-sequestration rate for recently deforested land

Assuming that the same linear growth rate is applicable to the recently deforested land in the study area as was used for the State of QLD (Chapter 9), but scaled to local site potential (using the benchmarks), then the re-sequestration of the emitted C from aboveground biomass in the study area would be 13,934 Mg $year^{-1}$, or a sequestration flux of 0.173 Mg ha^{-1} $year^{-1}$, or including root carbon, 0.230 Mg ha^{-1} $year^{-1}$. As in Chapter 9 the rate for the study area assumes no detriment to growth from climate change effects, such as increases in water deficit, storm and fire effects.

For re-sequestration of aboveground C by non-linear growth, equivalent to woody-thickening or slow plantations (Figure 10-9, Table 10-3), the value of *potentialC* used in Eq10-1 and Eq10-2 was the average (NCAS and ALOS): 21.05 Mg ha^{-1}. Sequestration with reforestation would mostly be at 0.009 (±0.002)–0.16 (±0.04) Mg ha^{-1} $year^{-1}$ for the first 10 years and 0.13 (±0.3)–0.32 (±0.08) Mg ha^{-1} $year^{-1}$ for the first 50 years, and full recovery would take at least 200 years (Table 10-3). The 50 year rate for the average equation is thus close to that for linear growth.

The destocking necessary for re-sequestration, based on the rain-dependent stocking rate formulae (Eq10-3 and Eq10-4) is: 30423 and 46742 DSE for the actual stocking rate and the notional carrying capacity, respectively. The corresponding avoided emissions from enteric fermentation upon destocking the deforested land are 739 Mg and 1136 Mg respectively. After 50 years that corresponds to avoided C effluxes of 0.03871 and 0.05948 Tg, respectively. The carbon cost for the livestock enteric fermentation emissions is approximately a tenth of the deforestation cost; and a carbon price of \$10 Mg^{-1} of CO_2-e equates to only ~\$0.93 DSE^{-1} $year^{-1}$, which is small compared with the grazing profit per average sheep of \$11.54 DSE^{-1} $year^{-1}$. Therefore the cost of livestock methane emissions alone is unlikely to motivate destocking.

Table 10-3. C sequestration and C income for different growth rates for reforestation of recent deforestation. Sequestration values are for aboveground biomass and income is for total biomass. Error margins in monetary incomes reflect likely variability in growth rates.

	C Sequestration				C income		
	Eq10-1			Eq10-2	Eq10-1, G=15 years	Eq10-2	Average
	G (years)						
	8	10	15				
	average flux						
	(Mg ha^{-1} yr^{-1})				(AUD$ ha^{-1} yr^{-1})		
1st 10 years	0.48	0.32	0.12	6.8x10^{-3}	5.8 (±1.3)	0.25 (±0.06)	3.1 (±2.7)
1st 50 years	0.31	0.29	0.24	0.10	12 (±5)	3.7 (±0.9)	8.2 (±3.3)
	total sequestration						
	(Tg)				(million AUD$)		
1st 10 years	0.41	0.27	0.10	5.8x10^{-3}	4.9(±1.1)	0.21(±0.05)	2.6(±2.3)
1st 50 years	1.3	1.2	1.0	0.42	49(±20)	15(±4)	35(±14)

After 10 years there was $3.51M and $5.39M profit ($4.5 (±0.9) M) for livestock grazing and $17.6M and $27.0M ($22 (±5) M) after 50 years (for actual and notional livestock capacities). Those grazing profits are greater than the C sequestration profits if the reforestation growth rate is that of woody-thickening (Eq10-2), but less than the C sequestration profits if growth is at the slow-plantation rate (Eq10-1, G=15) (Table 10-3). If growth is half–way in between those rates then grazing profits are greater than those from C sequestration after the first ten years but less than after 50 years. The uncertainty in reforestation growth rate therefore makes it unknown whether grazing or C sequestration is more profitable after 50 years.

10.4.4 Other re-sequestration rates

The same curves used for modelling reforestation of the recent deforestation were used for modelling the rehabilitation of woody biomass for the other levels of rangeland modification (Table 10-4). The profit for carbon replenishment was less than that from commercial grazing (for both the slow plantation growth rate and the thickening growth rate). Note however that the rates are spatial averages, and that they would initially be higher for denuded but otherwise intact land than for land with trees and shrubs. Selection of locally denuded land alone would thus provide higher rates of return. Rates will also be higher in the higher-rainfall zones, and if it is found that the NCAS biomass values are more realistic for remnant vegetation than the ALOS values.

Table 10-4. C sequestration and income for different forest degradation levels. Sequestration values are for aboveground biomass and income is for total biomass. Error margins in monetary incomes reflect likely variability in growth rates.

rangeland targeted		C Sequestration		C income	
		Eq10-1	Eq10-2	Eq10-1	Eq10-2
All		average flux			
		(Mg ha^{-1} yr^{-1})		(AUD$ ha^{-1} yr^{-1})	
	1st 10 years	0.48	$6.8x10^{-3}$	5.8 (±1.3)	0.25 (±0.06)
	1st 50 years	0.31	0.10	12 (±5)	3.7 (±0.9)
		total sequestration			
		(Tg)		(million AUD$)	
	1st 10 years	0.41	$5.8x10^{-3}$	4.9(±1.1)	0.21(±0.05)
	1st 50 years	1.3	0.42	49(±20)	15(±4)
Land below remnant value		average flux			
		(Mg ha^{-1} yr^{-1})		(AUD$ ha^{-1} yr^{-1})	
	1st 10 years	0.48	$6.8x10^{-3}$	5.8 (±1.3)	0.25 (±0.06)
	1st 50 years	0.31	0.10	12 (±5)	3.7 (±0.9)
		total sequestration			
		(Tg)		(million AUD$)	
	1st 10 years	0.41	$5.8x10^{-3}$	4.9(±1.1)	0.21(±0.05)
	1st 50 years	1.3	0.42	49(±20)	15(±4)
Land below extant average		average flux			
		(Mg ha^{-1} yr^{-1})		(AUD$ ha^{-1} yr^{-1})	
	1st 10 years	0.48	$6.8x10^{-3}$	5.8 (±1.3)	0.25 (±0.06)
	1st 50 years	0.31	0.10	12 (±5)	3.7 (±0.9)
		total sequestration			
		(Tg)		(million AUD$)	
	1st 10 years	0.41	$5.8x10^{-3}$	4.9(±1.1)	0.21(±0.05)
	1st 50 years	1.3	0.42	49(±20)	15(±4)

10.4.5 Cost of ongoing emissions

Apportioning of the Australian annual rangeland emission to NSW, with adjustment for the NSW environment, gave the average emission from deforestation over the last two years as 0.15036 Tg year^{-1} total C in biomass, and the carbon emission from Mulga lands forest degradation was 0.228 Tg year^{-1} (Table 10-5). The ΔSOC for NSW deforestation estimated pro-rata from QLD data was estimated to be 0.0147 Tg year^{-1}, and on a flat -17% basis over 30 years it was 0.0184 Tg year^{-1} SOC efflux: giving an average of 0.0176 Tg year^{-1}.

Table 10-5. Annual C emissions from NSW commercial rangelands. Where Australia-wide literature is cited, areas for NSW were determined in the present work and data applied pro-rata. Note that although SOC cost for deforestation is lower annually than biomass cost, it will continue for longer.

Activity producing emission	Emission data source	Tg year^{-1}	Cost (AUD$ million)	% of Total 1	% of Total 2
Mulga Lands degradation	Moore *et al*. (2001)	0.1768	6.48	32	18
Diesel consumption	Rolfe (2002)	0.01140	0.418	2.1	1.1
Soil erosion	Chapter 9	0.1406	5.15	26	14
Deforestation— biomass C	present work	0.1504	5.51	28	15
Deforestation— SOC	Chapter 9	0.01755	0.643	3.2	1.8
Total 1		0.5475	20.0	100	55
Livestock methane	Robertson (2003)	0.6783	24.9		68
Livestock methane	present work	0.2299	8.43		23
Total 2		1.001	36.7		100

The annual cost of maintaining the NSW rangelands under commercial grazing through current management methods is therefore, when accounting for C emissions, approximately $37M. The error margin for this figure is likely to be around 50 to 75%. After 50 years the cost of NSW commercial rangeland emissions is thus $1800 (±1100)M.

Introducing land rehabilitation State-wide for NSW in 2014 to counteract the ongoing soil erosion associated with commercial grazing and cessation of all that particular source of erosion by 2053, could limit the SOC efflux since onset of rangeland commercial grazing to 12 (±011) Tg (a cost of $434M), and emit only 5.1 (±4.6) Tg from 2014–2053. Thus, a SOC emission of 99.4-5.1= 94 (±41) Tg could be avoided by initiating rehabilitation in 2014 and ceasing anthropogenic erosion by 2053. That corresponds to an avoided cost of $3400M. The error margins for these estimates are high, possibly around 90%.

10.5 Discussion

10.5.1 Main findings and their wider applicability

It was found that over the study area as a whole and under current climatic conditions, farming profits exceeded the notional return from carbon re-sequestration, even if ongoing emissions were also priced. Nevertheless, spatial variations suggest that some localised destocking of the more-C-depleted areas would be profitable.

The complication of costs and carbon footprints of managed rehabilitation was circumvented in the present work by using 'natural' regeneration rates applicable to degraded rangeland, via modelling of woody thickening and slow-growing plantations. The approach focussed only on major effects and was robust, but the error margins were large. Finer-scale studies have reported sequestration profits, such as for ΔSOC with 'rotational grazing' (e.g. Gosnell et al., 2011) but they rarely include a full life-cycle C analysis of the intensive management, involving extra fencing, watering points and labour.

Our model of passive reforestation produced slower growth in the early stages compared with plantations (even-aged growth) (Figure 10-9, Table 10-3, Table 10-4)

because colonisation is likely to be patchy; infill may have to wait for seed production by the initial wave of colonisers (Chew and Chew, 1965), stand growth continues until the recruitment rate equals the mortality rate, and some later woody species may not germinate until the later stages of succession.

The linear sequestration rate assumed in most studies (e.g. Bray and Golden, 2008; Dwyer et al., 2009; Fensham and Guymer, 2009; Chapter 9) essentially equates to an average of Eq10-1 (with G=15) and Eq10-2; faster than woody-thickening, but slower than plantations. Linear rates exaggerate sequestration in the early years. The average rate for the first ten years (Table 10-3) is similar to that proposed for the Henbury property in the Northern Territory (Australia) (DSEWPC, 2011), which has similar rainfall to NSW (293 mm $year^{-1}$ and 320 mm $year^{-1}$, respectively).

Re-sequestration rates will be spatially variable. Areas with species with lignotubers (e.g. mallee eucalypts and brigalow) will regenerate faster (e.g. like Eq10-1 with G=15 years), but sites that have been repeatedly deforested or undergone substantial soil erosion will be slower, similar to woody thickening (Eq10-2), though after a century sequestration is similar. Rates will be higher in the less water-limited regions, but so too will livestock profits.

NSW rangelands contain three global biomes, which should allow the NSW re-sequestration rates to be applied to areas outside of Australia with similar biomes and climates (Figure 10-15, Table A-V-8,9, Figure A-V-17) as initial estimates, although it is strongly recommend that differences in biotic and abiotic conditions be taken into account if the methodology presented here is applied in other countries.

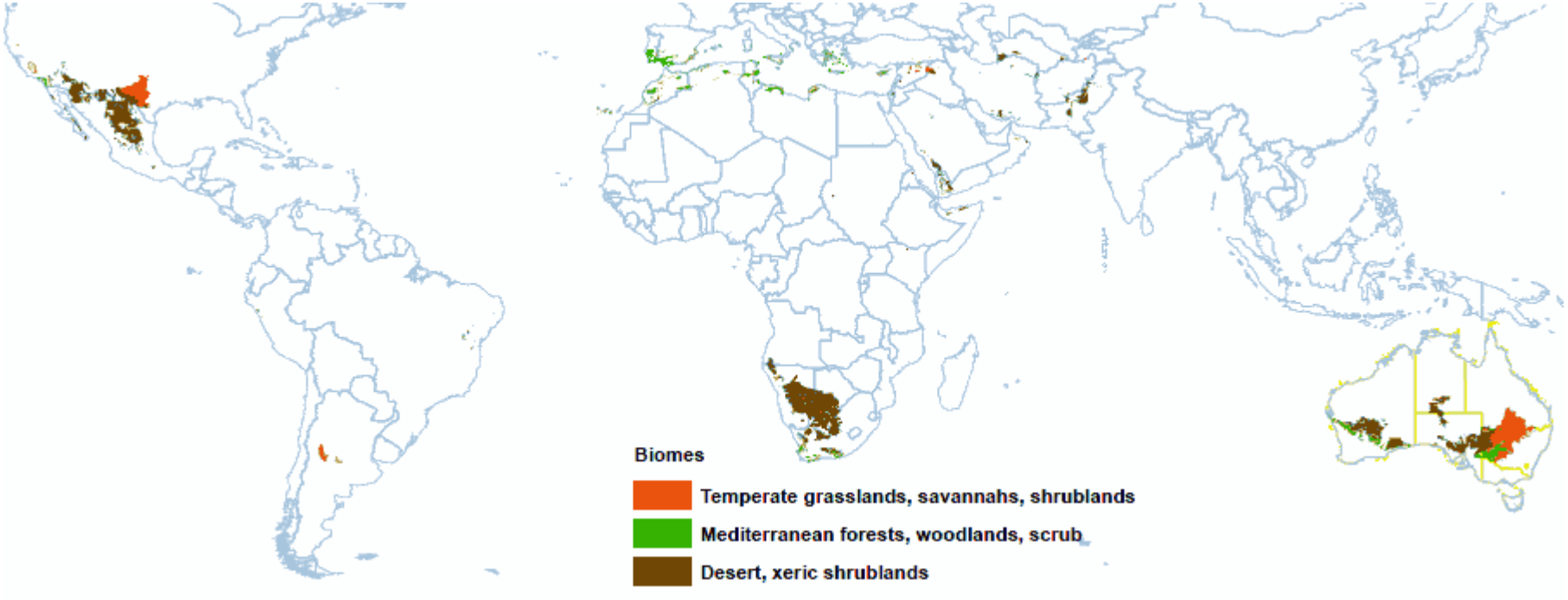


Figure 10-15. Global rangelands with the same biomes and similar climate (rainfall and average temperature) to those of NSW's rangelands. Close-up maps in Figure A-V-17.

10.5.2 Recent deforestation

For emission calculations in the present work, it was assumed that the C is emitted spontaneously upon clearing, which is not so even under the common practice of heaping and burning debris (Dwyer et al., 2009), and with roots less likely to burn. If trees and shrubs are left to decompose after felling, C loss can take over a century. Positive feedback from climate change (Bonan, 2008) which can be nonlinear (Raupach et al., 2011; Raupach et al., 2014) means that the timing of emissions may contribute to their effect. The black carbon and methane from burning may produce an effect additional to the CO_2 emission alone (M. Raupach, personal communication, 2014) which under full costing, would incur a higher penalty than aerobic decomposition. After felling, decomposition emissions can be reduced by sealed deep burial (e.g. Zeng, 2008) as can fire hazard. The balance between the two post-felling activities will also be influenced by residual charcoal, how much tree carbon becomes SOC, how much decomposition is aerobic (as opposed to anaerobic [methane] emissions), and the ecological functioning of the surface debris. Such processes are lengthy and so it would be more feasible to recover emission costs at deforestation time: for example by applying fees for lost biomass, associated ΔSOC and any positive feedback.

Both present and future livestock densities and carbon prices are important when considering land use alternatives. With actual and notional stocking rates differing by ~29% and the sequestration profitability determined by the choice of regrowth formula, it is obvious that the real stocking rate has a strong influence on outcomes. A complication in assessment is that stocking rates can usually be maintained high for a couple of decades after deforestation, while the nutrients from decomposing biomass and the legacy nutrients are available, after which stocking rate must drop (Sangha et al., 2005; Kaur et al., 2006; Radford et al., 2007; Kirschbaum et al., 2008).

The potential reforestation C flux of 0.13 (±0.3)–0.32 (±0.08) Mg ha^{-1} $year^{-1}$ (including roots) is similar to the 0.22 (±0.02) Mg ha^{-1} $year^{-1}$ (excluding roots) of Witt et al. (2011) for degraded Mulga forest in QLD, but less than the 0.36 (±18) Mg ha^{-1} $year^{-1}$ for QLD (Dean et al., 2012c) but the QLD rangeland included more high-productivity land and the remnant ecosystems may have been less degraded.

The methane emissions avoided by destocking were comparable with the sequestration in biomass for the slowest growth rates. However, there are other C benefits, such as avoiding typical SOC emissions resulting from grazing. For stronger climate change mitigation, a key part of carbon management is conservation of carbon stocks, that is to forgo the deforestation, but the current market and policy implementations are inadequate (e.g. Krause et al., 2013). One study found that objections to deforestation controls were economic, though a carbon pollution cost for deforestation was not included (Davidson et al., 2006).

10.5.3 Major biomass change for different levels of forest degradation

The major re-sequestration options calculated here vary considerably in their C totals and hence potential financial dividends (Table 10-1). The estimates of 34.5 and 35.5 Tg from ALOS and NOAA-AVHRR processed data respectively for NSW-wide re-sequestration can be compared with 42.3 Tg from Chapter 9. The values calculated in this chapter used refined rangeland boundaries and should therefore be more spatially representative. An apparent paradox is that, on average, the entire NSW rangelands can re-sequester 35 (±10) Tg whereas tallying only the land below-remnant-ecosystem-biomass can re-sequester 61 Tg. This arises from using two different summation methods. In the former, land that was already above potential was tallied together with land that was below potential (over wide expanses), i.e. allowing additions and subtractions to partially cancel each other. Although not shown in Table 10-1, if only land below-potential was considered NSW-wide then 81 Tg could be re-sequestered.

The reasons for merging above- and below-potential land were: (a) to smooth over noisy data, and (b) to remain conservative in the calculation of monetary dividends for re-sequestration and thus avoid disappointing speculation. The 35 (±10) Tg NSW-wide is also conservative because reliance on depauperate remnants as indicators of potential gives a false indication that more rangeland is already above its potential carbon stock than it really is— these areas then mathematically cancel out the more severe C deficits upon averaging, to give an overall low re-sequestration forecast.

The low biomass values from ALOS data for ecosystem remnants (compared with NCAS or NOAA-AVHRR values) (Figure A-V-12) can be attributed to biomass attrition with commercialisation (Allen, 1983) and to the NCAS formulation not accounting for native grassland (Figure A-V-4). Some entire remnant ecosystems had substantial biomass attrition (Figure 10-4), especially those neighbouring riparian zones (e.g. 'Mid-Darling Plains' between Cobar and Bourke, alongside the Darling River, Figure 10-14). The concentration of attrition in riparian forests in rangelands is Australia-wide (Chapter 9) and reveals the need for more-adequate benchmarks for broad-scale carbon assessment schemes.

Conversely, ALOS values were higher than NCAS values in the few high-rainfall areas, which have taller forest (Figure A-V-12). That may be due to the tendency of NCAS to underestimate biomass for tall native forests (Keith et al., 2010) or for radar to saturate in some higher-biomass woodlands (Lucas et al., 2000).

10.5.4 Climate change effects

The effects of climate change, and accompanying changes in atmospheric chemistry, upon vegetation and land condition (and thus on rehabilitation), depend on a range of climate attributes and biotic responses, and on feedbacks between these. In summary, the positive influences of CO_2 and N fertilisation are overshadowed by the multitude

of secondary negative effects such as increased fire and seasonal stressors (Chapter 9). Arguably the best estimate of change in biomass due to climate change over the coming century comes from an ecologically based calculation of ΔSOC. The relationship is reflected in the correlations found here between biomass and rainfall, SOC and temperature, and livestock carrying capacity and rainfall (Eq10-3, Eq10-4). Forecast ΔSOC to 0.3 m depth, from 2000 to 2100 for NSW rangeland, for a range of climate change scenarios, averaged -27 (±11)% or a SOC emission of 163 (±66) Tg (Chapter 9), equating to AUD$6000 (±2400) M. That modelling was based on the relationship between rainfall, temperature, biomass and SOC, and inherently includes tributary characteristics such as drought and fire, which are expected to have an increased influence with climate change (Gonzalez et al., 2010). The 58 (±24) Tg possible C re-sequestration (for biomass) for NSW, calculated here, was in the absence of climate change and may thus need to be reduced by up to ~27%. The effect of climate on ΔSOC is unlikely to be expressed within one century— it will take longer and be largely in response to change in biomass over that period, i.e. ΔSOC will be delayed compared with the change in biomass.

The temperature and rainfall extremes of climate change are direct threats, influencing net growth (e.g. Wallace et al., 2009; Piao et al., 2011b). If ecosystems cannot adapt fast enough to the new climatic extremes then biomass will decrease more than forecast by change in annual rainfall and temperature averages alone. A study of climate change impacts for Namibia (which has a similar xeric biome to that of NSW, Figure A-V-17, Table A-V-9) found that with decreasing rainfall and increasing temperature (as are forecast for most of NSW), there was a significant decrease in tree biomass when change in fire regimes was included (Sarker et al., 2012). Most climate-change-impact studies neglect change in fire regimes. Many also neglect the increase in decomposition rates that has been forecast to offset CO_2 fertilisation (Trumbore, 1997). A climate-change study for Morocco (Schilling et al., 2012) which has a significant proportion of rangeland similar to the Mediterranean-biome in NSW, showed similar climate-change effects to those for NSW. The study highlighted a forecast reduction in agricultural production and the need for conservation of productive assets. The positive correlation between livestock

carrying capacity and rainfall found by us for NSW suggest a similar decrease in productivity.

The largest single threat to Australian rangeland C sequestration is an increase in fire, and it is forecast to increase with climate change and with population. Fires that are too frequent reduce site-quality and long-term carbon storage, and may increase erosion (e.g. Janzen, 2011; Shi et al., 2013). Conversely, decreased fire frequency is correlated with decreased grass density and conversion of savannah and woodland to forest (Anella and Wright, 2004; Dutilly-Diane et al., 2006; Fosse et al., 2008). Grass fires can offset increase in woody biomass following good rains, eliminate much coarse woody debris and cause increases in fire-tolerant invasive species such as buffel grass (*Cenchrus ciliaris* L. (syn. *Pennisetum ciliare* (L.) Link)). The negative impact of buffel grass on woody biomass has been discussed previously (Yates et al., 2000; Chapter 9; Butler and Fairfax, 2003; Dwyer et al., 2009). It will become more of a problem for NSW with climate change because it will follow the wave of suitable climatic conditions from the continental interior southwards and south-easterly (Steel et al., 2008; Martin et al., 2013). Biological control methods for buffel grass might be effective in reducing its threat.

10.5.5 Cost of ongoing emissions

With a carbon price of AUD$10 Mg^{-1} CO2-e, after 50 years the cost of NSW commercial rangeland emissions is $1800 (±1100)M. That is of similar magnitude to the income of $1200 (±480)M after 50 years from rehabilitating the 17.90 Mha of land that has carbon stocks below the 'remnant' values. These are opposite management alternatives, but if sequestration is chosen over grazing then there is both an avoided cost (of ongoing emissions) and a monetary dividend (from re-sequestration in vegetation), making a difference of $3027M, if the same market place is used for both transactions. That type of balance would also apply at the property-level. Alternatively, the annual NSW emissions cost of $36.7M is ~35% of the NSW farm net profits of $104.3M. Thus, on average, the emission costs of

rangeland farmers could be recovered by a fee of 35% of their net profits, and used for managed reforestation of abandoned land.

The estimates of emissions from livestock were significantly lower than those from previous work (viz. Robertson, 2003). The livestock numbers for NSW are a possibly more-direct estimate (relying on NSW State data) rather than the result of dividing the nationwide estimate of methane emission according to land area, pro-rata, and the methane emission per DSE was conservative. Slightly counteracting this conservatism was the assumption that destocking would not substantially increase feral goat populations, using an assumption of decommissioning artificial waters and some continuance of goat harvesting.

NSW-wide, the two-highest singular annual emissions were forest degradation in the Mulga Lands and methane from livestock. Conversely the flux of SOC emissions accompanying deforestation was low but continues for one or two millennia, due to a longer average half-life of SOC. Additionally one must note that deforestation produces a positive spike in SOC stock for up to 15 years, which experimentation and any C-trade assessment must accommodate Chapter 8.

The SOC efflux for NSW due to erosion induced by commercialisation (derived pro-rata from the Australia-wide estimate in Chapter 9) is 7.1 (±1.8) Tg. SOC takes an order of magnitude longer to sequester than biomass carbon, and land rehabilitation is an additional time requirement; therefore the 7.1 Tg is likely to take several centuries to re-sequester. Despite the financial irrelevance of such a timeline in current markets, rehabilitation also avoids the C effluxes of continued grazing. The NSW-wide annual costs for current SOC losses from erosion and deforestation are $5.2M and $0.62M, respectively (Table 10-5). Measuring ΔSOC appropriately is paramount, though very few experiments have sampled the dynamics of SOC over the whole soil profile, or over sufficient time, and accounted for the lag of ΔSOC behind change in woody biomass (Shi et al., 2013; Chapter 8). Thus the monetary value of ΔSOC is unlikely to be realised by financially focussed management and thus change in current practices would require alternative motivations.

10.5.6 Increasing certainty

Defining the rangelands by a single boundary is an administrative aid, whereas in reality land uses are variegated either side of that line, as for most agricultural boundaries (Whittlesey, 1936). Classification based on land use may also be ambiguous. For example, where the native woody vegetation has been kept cleared such that it cannot regenerate autonomously, and the grasses in its place are exotics, then the land could be considered intensive agriculture (e.g. Figure 10-5, Figure 10-7.a, Figure 10-8). Overall, 'rangeland-ness' needs to be better-determined for rangeland carbon accounting.

Potential biomass defined by remnant vegetation was most probably underestimated, due to almost ubiquitous forest degradation and because farmers usually deforest the most-productive land first (Pressey et al., 1996; Holmes et al., 2006). The strongest effects of this selective process were near major rivers, and in the arid west where there was woody biomass attrition without broad-scale deforestation. Also, rivers with high biomass from overhanging *Eucalypts camaldulensis*, were included in the extant 'average' but were absent from the 'potential' because they were not flagged as native ecosystems [as their remotely sensed pixels had the possibility of being water-dominated]. Better, though more time-consuming representation of potential could possibly be obtained by detailed modelling of ecosystems, with interpolation over deforested and other highly degraded areas using software such as ANUClim (Xu and Hutchinson, 2014) along with ground-truthing and use of historical aerial photography.

Ideally, stocking levels would be available in the form of annual averages over several years, because land condition responds cumulatively to long-term effects. This would allow determination of a more robust correlation between stocking levels and environmental variables. Nevertheless, possibly due to the large size of NSW, a degree of correlation was noted. The paucity of paddock-level stocking data for Australia, and apparent covertness or passive-non-compliance by farmers and some

government departments, has been noted previously (Bartel and Barclay, 2011; Chapter 9) and remains a hurdle to productive environmental management and scientific progress.

The north-south gradation in temperature in NSW contrasts with the east-west gradation in rainfall (Figure A-V-5.a, b). This produces a greater range of ecosystem types than if the two gradients were parallel, and thus a greater variety of knowledge is required to develop C trading scheme 'methodologies'. Areas found to be depleted in the present study are guides only— on-ground verification is advised prior to investment.

The high error margins for re-sequestration were partly due to the use of broad-scale averages and because of spatial variability in forest degradation level. Most individual land-use polygons (the area unit of calculation) spanned a mixture of barren areas, degraded forest and vestigial trees or shrubs. Sequestration will primarily be by infill. Within the barren patches sequestration rates will be similar to those for forest-stand growth, but for the larger polygons it will necessarily be less. Thus, in practical terms for greater certainty in financial sequestration projects, fine-scale vegetation mapping is required and different growth rates assigned to canopies and canopy gaps. Also, the high error margins noted for sequestration over the first decade cannot be reduced without more knowledge about the dynamics of autonomous reforestation.

10.5.7 Wider implications

The modelling in the present work indicate that there may be some circumstances in which destocking for carbon gives a better return than maintaining stock grazing. There are environmental, social, cultural and other economic consequences of any resultant shifts in land use. Multiplier effects from carbon payments may result in different outcomes than those from farming activities. While it is expected that destocking will result in greater availability for commercial harvesting of feral and

native animals (even when artificial watering points are decommissioned), thereby mitigating the impact on re-sequestration of any wild animal population increases, the employment types resulting from carbon management and altered protein sourcing are likely to be very different from those of the present, with likely positive and negative social and cultural consequences. In many areas where soil and the native seedbank have been severely degraded (Brown and Potter, 1971; Condon, 1986; Sparrow et al., 2003; Hunt et al., 2014), investment in rehabilitation is likely to be necessary to achieve the re-sequestration forecasts calculated here. The achievement of maximum carbon re-sequestration in the landscape is not necessarily the best nature conservation outcome in the short term because maturation of woody-thickening (accompanied by self-thinning) may take well over a century (Hibbard et al., 2003; Chapter 8), and meanwhile woody-thickening is regarded as degradation in the NSW regulatory processes for land clearance.

In a much broader context, the results also raise the need for the substitution of protein derived from any destocked rangelands by protein from other sources (to avoid carbon emission leakage by mere transferral of activity, Tamminen (2011)). At a global level, options include the increased use of vegetable sources and a reduction in food wastage (Wirsenius et al., 2010; Ripple et al., 2014). In the longer term, tissue-cultured protein may provide a solution (e.g. Edelman et al., 2005; Hopkins and Dacey, 2008). Regarding leakage for wool textile production, alternatives are available, such as organic cotton and flax. Detailed regional greenhouse footprints for textile production are rare but analyses for wool production without incorporating the LUC contribution yields ~22 kg CO_2-e/kg wool (Brock et al., 2013; Wiedemann et al., 2016) whereas comparable analyses for flax (*Linum usitatissimum L.*) yielded ~0.80 kg CO_2-e/kg flax (Barth and Carus, 2015), and for dry-land cotton yielded ~1.7 kg CO_2-e/kg cotton (Maraseni et al., 2010).

In summary, while the present study has shown how information and certainty on the spatial concentration and fluxes of carbon in rangelands can be increased, and has indicated the potential for use of some land for carbon re-sequestration, there is insufficient detail for carbon trading. For financial sequestration projects, vegetation

mapping would be required at a scale where individual tree canopies and bare ground are identifiable. Knowledge of long-term ΔSOC would also greatly improve accounts of national emissions. To increase precision in both emissions and sequestration values, experiments are needed to increase knowledge of:

(a) allometrics for trees and shrubs including deep-set roots (that access ground water),
(b) natural regeneration growth rates,
(c) the timelines involved in SOC formation, mobilisation and decomposition for different ecosystems
(d) changes in tree carbon stocks under grazing, and
(e) the locations of representative remnant ecosystems, with at least 90% of pre-commercialisation biomass present.

Despite the low returns from C sequestration calculated from NSW-wide averages, the accumulated debts from ongoing emissions are high, even at the middling carbon price of AUD$10 Mg^{-1} of CO_2–e. At that carbon price, and a hypothetical situation of "polluter pays", the current annual net emissions from the Australian rangelands equate to AUD$0.74 (±0.4) billion, or AUD$1.1 (±0.5) billion including methane from livestock and savannah burning.

With global climate change the emissions from rangelands will increase, with the best regional estimate coming from the change in soil carbon (down to 0.3 metres). For the Australian forested rangelands SOC emissions initiated by climate change to 2100 (though completed over several centuries) will be 1130 Tg (Chapter 8), equating to AUD$41 billion, which is the largest emission calculated in the present study. The similar debt associated with carbon in lost biomass will occur sooner.

Chapter 11 Discussion

11.1 Questions

The aim of the thesis was to work towards solving the most substantial but probably solvable gaps in knowledge about the influence of human land use on carbon dynamics in woody ecosystems. The type of knowledge sought was quantitative and explanatory. For convenience the study focussed on Australia but it was global in terms of literature, data, concepts, applicability and effects. The concepts therefore needed to be basic, widely relevant and pertinent to current carbon management.

The present work is necessarily part of ongoing purposeful activity worldwide—providing a foundation for addressing carbon emissions and sequestration associated with woody ecosystems for commercial and other management ventures. The higher-level concepts developed here for particular ecosystems can be transferred to other ecosystem types and carbon management elsewhere and are thus relevant to a wide audience. The findings will then contribute towards more precise estimates of carbon fluxes for emission trading schemes, climate change modelling, and national reports.

The work in this thesis has met its aims and contributed to current knowledge in four main areas:

1) increased formulaic relations between natural phenomena for use in modelling carbon dynamics,
   (35 new equations were derived: 19 for trees or C in stand biomass, 14 for SOC, and 2 relating land use to climate; 9 of the SOC formulae had vegetation attributes as independent variables. The equations ranged from species-specific to widely applicable.)
2) showing how a time-space (4D) context of carbon dynamics in woody ecosystems can improve understanding of experimental result
   (e.g. woody-thickening 'break-even' time; more SOC loss with deforestation than previously estimated; increase in wood-product half-lives necessary to

regain primary forest carbon stocks; and long-term SOC loss with intensive logging and harvesting cycles);

3) increased understanding of soil organic carbon distribution and change (e.g. development of 'Method 4' to get cumulative SOC with depth, thereby allowing representation of a diverse soil profile and the SOC associated with large trees which have been targeted for anthropogenic use); and
4) increased quantification of anthropogenic influence on C flux in woody ecosystems (emissions with logging and with rangeland commercial grazing, past and future).

The first part of the thesis was the fine detail of the more carbon-dense and intensively used wooded ecosystems— the tall open-forests (TOFs)— where biomass usage and accompanying disturbance, mostly by clearfell logging, is intense; the second part was more macroscopic, on the larger areas of the less carbon-dense and less intensively used wooded ecosystems— the 'extensive' rangelands— where a higher percentage of available biomass is converted to emissions on site.

The initial work aimed to show connections between ecology, C-distribution, timber extraction, climate change and climate change mitigation, for primary forests (Chapter 2). The aim of the next chapter was to derive a stand-level estimate of C for a specific forest type and in more detail than earlier work. It including theoretical development of allometrics for large *E. delegatensis* and generic rainforest understorey, and a carbon emission forecast from recent logging of that forest type.

The question arose as to which recent logging, of 'wet' and carbon-dense or of the 'dry' and more numerous forests, had generated the highest emissions and just how far the targeting of denser forest for wood extraction had been manifested (Chapter 4). The corollary was to determine if emissions from that major usage are recoverable from the residual C in wood-products or how the emissions could be ameliorated while maintaining wood supply.

Due to the spatial concentration of biomass carbon in large trees and the literature suggesting that SOC should be more concentrated closer to them, the aim of Chapters 5 and 6 was to assess the influence of SOC near large trees on typical estimates of SOC per unit area.

The aim of Chapter 7 was to solve a pervasive controversy regarding the fate of forest soil carbon under timber production. The aim of the ensuing chapter was similar but applied to rangeland: to help to settle the conflict on the direction and magnitude of change in rangeland soil organic carbon following changes in woody cover.

The aim of Chapter 9 was to determine magnitudes, timescales, knowledge gaps and management options for major carbon fluxes for the commercial rangelands of Australia with change of woody cover. The re-sequestration example was one where baselines (being the remnant ecosystems) were well defined. This chapter, like Chapter 2, aimed to determine the effect of climate change on SOC, but for carbon-sparse woody ecosystems.

As Chapter 4 followed Chapter 3 with more-detailed C modelling for carbon-dense forests, so Chapter 10 followed Chapter 9 with more detail for carbon-sparse rangelands. The aim was to determine a method for locating the most lucrative places in rangeland for C re-sequestration, and required:

1) derivation of formulae for C re-sequestration rates,
2) derivation of a method for defining potential magnitude where ecosystem remnants were poorly defined, and
3) recognising the relevance of the findings to similar rangelands elsewhere.

11.2 Specific contributions to current knowledge

The work in this thesis has made specific contributions in the following areas:

- Knowledge was improved on the dimensional needs for empirical detection of change in carbon stock. It provided a more-comprehensive vantage point from which contraindication from results (as apparent in interpretations presented in the scientific literature) disappeared, for both forest logging, and rangeland woody-thickening and deforestation. For clearer interpretation of different reports on forest logging, it was found necessary to differentiate trees at different stages of maturity, namely those over 100 years as a group, from those over 200-to-250 years. Similarly, in the full-life-cycle analysis of LUC, product substitution is usually not treated as a fully quantitative variable but nevertheless it has a spectrum of values, for which calculations must consider a wider range. Products with a lower carbon footprint than timber must also be considered, not just those with a higher carbon footprint. The rangeland equivalent is that the capacity must be determined (financial, practical, feasible, and in terms of C flux) for the *in vitro* meat industry as a substitute source of protein to replace commercial rangeland grazing. Lengthening of timespans for empirical experiments to match half-lives of SOC was found necessary for LUC to both rangeland grazing and forest timber extraction. The forecast total carbon of forest subject to logging must consider the historical primary forest; and accounting for forest soil carbon must consider multiple harvesting cycles. To determine SOC change for rangelands the SOC forecast for woody-thickening must allow time for maturation and decomposition of new woody stands. Calculation of SOC change accompanying deforestation in rangeland must consider environmental dependence of initial SOC stock.

- The carbon stocks and fluxes for some significant but previously poorly measured woody ecosystems and their anthropogenic usage were determined. For the carbon-dense forests, details were determined for the carbon stock distribution within a primary mixed-forest of the types that have been extensively logged, yet those types were not the most carbon-dense examples within that region's logging history. The findings detailed the intervention's

effect on the magnitude of carbon stocks, which stocks were most affected, and over what temporal scale. For rangeland the magnitude and location of altered stocks was determined, and a temporal scale and magnitude for the carbon flux of their recovery.

- Carbon allometrics were developed to include mature trees and formulae were developed relating SOC to these trees and to vegetation more generally. Allometrics are part of the mathematical foundation of carbon accounting, which in turn is part of the foundation of climate change modelling. Allometrics are also used in commercial re-sequestration or emission-avoidance projects. Mature-tree allometrics are necessary in determining what solid-phase C humanity has historically relocated; how much from the remnant forests humanity may alter in the future; the potential of re-sequestration projects; and as shown here, SOC stocks in difficult-to-measure places. Even if anthropogenic climate change ceased, due to the ubiquitousness of people interacting with parts of nature that constitute the global commons, it is necessary to know their impact on their surroundings, such as redistribution of its components. For carbon-dense forests this project improved allometrics for *E. regnans* in the buttress region, *E. delegatensis* aboveground carbon, and to a lesser degree for the generic rainforest understorey of mixed-forest. Related major points were: for carbon-dense primary forests half of the eucalypt biomass was from trees with DBH>2.4(0.1) m; large trees provide an extra 5% in SOC (66% of 6.9%) than previously measured in addition to high SOC in between them; the short-term emissions upon LUC are twice as high from mature primary-forest than young primary-forest; and the majority of LUC emissions over the last 20 years in Tasmania came from clearfelling mature stands of large trees. Not only is the carbon store disproportionately higher for larger trees but so too is the annual growth rate per mass of tree (Sillett et al., 2010; Stephenson et al., 2014). These facts indicate the importance of allometrics for large trees, even though there are fewer of them and their numbers are decreasing. Protection,

leading to conservation of the larger trees, should therefore help delay climate change impacts and delay positive feedback. Specific-tree allometrics were not determined for rangeland species but the potential carbon stock of specific ecosystems was determined, for NSW.

- Regarding the fate of carbon with primary-forest logging, there is a substantial long-term C emission for the LUC of primary-forest to harvesting cycles. It is redeemable with sufficient wood-product recycling or reducing mill losses to extend half-lives (including that in the paper stream) but practically it would be difficult to achieve. Unfortunately that cannot be applied to timber or forest stands that are already well down the usage stream, but it is more applicable to future LUC and its wood-products. Similarly, product substitution for wood products, without C leakage (i.e. in a stable wood market) is relevant to future LUC prevention. An additional attenuation of LUC emissions may come from SOC in landfill from decomposing wood-products under some circumstances: the total LUC emissions could possibly be kept under 50% of the primary forest carbon stock. Longer harvest cycles reduce emissions from LUC and have the benefit of engendering rainforest understorey, which may act to keep global average temperature lower and reduce the fire risk associate with climate change.

- A previously unmeasured SOC pool was quantified. When the under-trunk SOC was combined with organic soils associated with the buttress region and in nearly decomposed logs, SOC at the unit-area-level increased by ~7(3–12)% relative to the inter-tree SOC alone.

- State-level and national rangeland carbon emissions were determined, and C re-sequestration rates were determined for NSW and QLD. Climate change may hinder both re-sequestration and the grazing industry.

- C emissions accompanying LUC accumulate and have momentum for both forest logging and rangeland commercial grazing. The endpoint is several human generations away from LUC instigation. The emissions associated with the endpoint could be brought to the attention of the instigators. Both industries also have a legacy of large carbon emissions and a legacy of discord over C balance in the literature, and environmental impact more generally.

- Long-term net emissions forecast to arise from SOC due to climate change are about 100 times higher from the carbon-sparse Australian forested rangeland than from the Tasmanian rainforest, due to the area difference. LUC in TOF in Tasmania, from primary-forest to cyclically harvested forest will initiate equivalent emissions to climate-change-initiated long-term losses from Tasmanian rainforest. If the inevitable increase in fires in the west of Tasmania spreads east into the mixed-forest due to the dominant westerly wind direction, then one could easily envisage that the C stocks of the carbon-dense mixed-forest will further decline. This would still not make Tasmanian losses higher than mainland Australian rangeland losses, due to rangeland size.

11.3 Relation between land-use, woody vegetation and carbon emissions

The carbon cycle highlighting land-use, woody ecosystems and climate is shown diagrammatically in Fig. 11-1. (Atmosphere-ocean effects are implicit in the carbon efflux and climate relations.) A similar series of relationships is operational in the Amazon rainforest (UNEP, 2011 Figure 5) for which physical events were labelled explicitly; for example increase in fire intensity and frequency, reduced precipitation and warmer sea surface temperatures. In the global system there are too many

different events to draw in one such a diagram, therefore in Figure 11-1 they are grouped into net global effects and are discussed below, in terms of findings from the present work and those in the literature. There are exceptions to each of the relationships portrayed and therefore none are hard and fast rules.

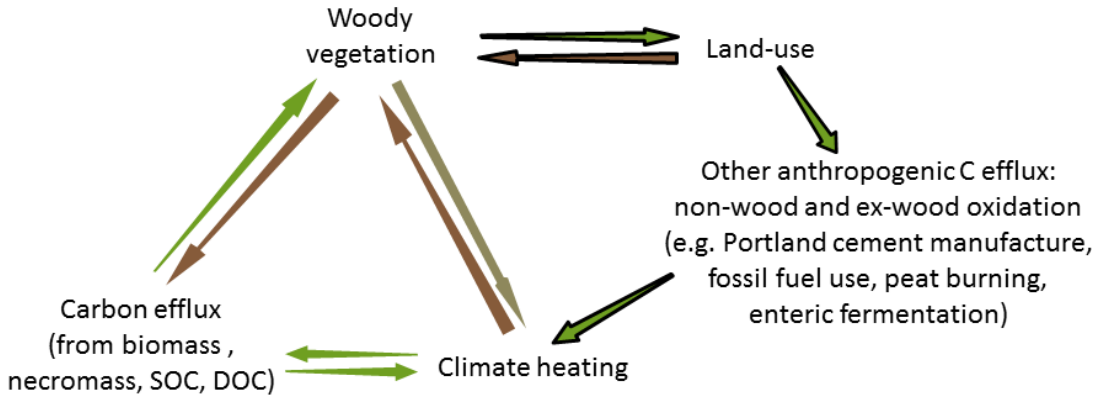


Figure 11-1. Schematic of within-planet, non-geological, net, global, surface carbon influences. Darker arrows: anthropogenic. Lighter arrows: anthropogenic and non-human. When an arrow widens towards its point that means an increase in the origin entity will increase the end entity (i.e. a positive correlation), whereas a narrowing arrow means a negative correlation. No change in arrow thickness means no as yet discernable net global effect (e.g. 'Woody vegetation' effect on 'Climate heating').

Positive feedback is indicated where adjoining entities are related by two positive arrows, e.g. between climate heating and biomass emissions. Feedbacks may also exist within entities or between entities that are indirectly connected. For example elevated CO_2 (from carbon efflux in biomass, SOC or DOC) may induce emissions directly by forming a higher portion of labile SOC (Hofmockel et al., 2011), with the majority of such inducing emissions presently coming from anthropogenic activity.

More woody vegetation intensifies land-use. Some examples are: (a) high-grading of Tasmanian forest over the last twenty years— more wet-eucalypt was logged than dry-eucalypt and more of the remnant areas of higher-biomass wet-eucalypt species were logged (Chapter 4), (b) clearfell is more likely in wet-eucalypt (high biomass)

while selective logging is more likely in dry-eucalypt (lower biomass), (c) global intensification of eucalypt plantations with shorter harvesting cycles in faster-growing locations (Hart and Santos, 2015), (d) higher livestock stocking rates in more-productive NSW rangeland, and (e) more rangeland degradation closer to rivers (which normally stock higher-biomass tree stands) in NSW rangeland.

Increase in woody vegetation within a locality decreases [net] carbon efflux. This is direct, through carbon sequestration in biomass (alive and dead), and indirect from higher biomass sequestration to increased SOC and DOC, e.g. maturity of large trees in primary forests, higher SOC concentration near larger trees, SOC in late-stage CWD, and rangeland woody thickening increases SOC in the long term.

More areas of woody vegetation may or may not increase climate heating. Globally it is a mixture of effects such as decreased albedo (heating) and evapotranspiration (cooling) and different effects in different locations (Bonan, 2008; Mahmood et al., 2014). Vegetation decrease through land management, without change in cover type has similar effects to change in cover type, on climate heating (Luyssaert et al., 2014), e.g. land cover going from forest to forest such as in conversion of primary *E. regnans* mixed-forest to harvesting cycles.

Decreasing woody vegetation increases carbon efflux, e.g. directly through burning or slow decomposition of killed trees (during conversion of primary forest to harvesting cycles, rangeland deforestation), delayed efflux from emissions during timber processing, shrub attrition from soil erosion in rangeland; long-term SOC emissions from primary forest conversion to harvesting cycles (where the temporal average biomass is reduced), and long-term SOC emissions following reduced woody vegetation in rangeland. Conversion of primary forest to cultivation-types of agriculture has a more pronounced reduction in SOC (i.e. a greater carbon emission) in the longer-term than in the short-term (Wei et al., 2014).

More land-use decreases woody vegetation, e.g. conversion of primary forest to harvesting cycles, rangeland erosion from overgrazing causing tree attrition, rangeland deforestation and fence-line clearing for increased livestock grazing, clearing and seeding with buffel grass causing decrease in rangeland woody vegetation. The exception of woody-thickening that often accompanies excess overgrazing of rangeland (with concomitant attrition of mature woody vegetation) can be viewed as a temporary phase of recolonisation, which undergoes slow self-thinning and that will eventually settle into a new equilibrium once the land is too depleted for that continued concentration of livestock. That new equilibrium is likely to once again attract high-intensity land-use.

More land-use increases non-wood and ex-wood (i.e. what was once wood) carbon emissions through infrastructure construction, use of farm machinery, product transport, and livestock enteric fermentation.

Less land-use increases vegetation, e.g. less frequent harvesting cycles allows more maturity in trees and thus higher biomass with likely higher forest SOC, rangeland woody and herbaceous regrowth with destocking (increasing biomass directly and SOC in the long-term).

Climate heating decreases vegetation on net globally, through longer summers, more droughts, increased severe-fire frequency. A contemporary exception is woody vegetation increase in the sub-polar zones (Bonan, 2008). Data available in the last decade provides increasing evidence of stress effects on forests [from climate change] either limiting growth or reducing standing biomass in many non-polar regions (e.g. Barber et al., 2000; Calder and Kirkpatrick, 2008; Allen et al., 2009; Wallace et al., 2009; Carnicer et al., 2011; Piao et al., 2011a) through increased severity or frequency of droughts and fire.

Climate heating increases carbon efflux, through increased frequency of severe fires (Liu et al., 2010; Holz and Veblen, 2011; King et al., 2011; Fox-Hughes et al., 2014).

In the longer-term this will change ecosystems to those of lower-biomass, possibly to ones more-acclimatised to fire. Either way it will produce a long-term net efflux from both biomass and SOC, as forecast for southwest WA and Tasmania. At the microbial-level, within certain temperature and moisture ranges, higher temperatures increase decomposition of biomass (assuming ample water and air availability), necromass or SOC, i.e. increased carbon efflux with possible positive feedback (Kirschbaum, 2004). However a range of ecological processes must also be considered to quantify the longer term effects (Crowther et al., 2015; Bradford et al., 2016).

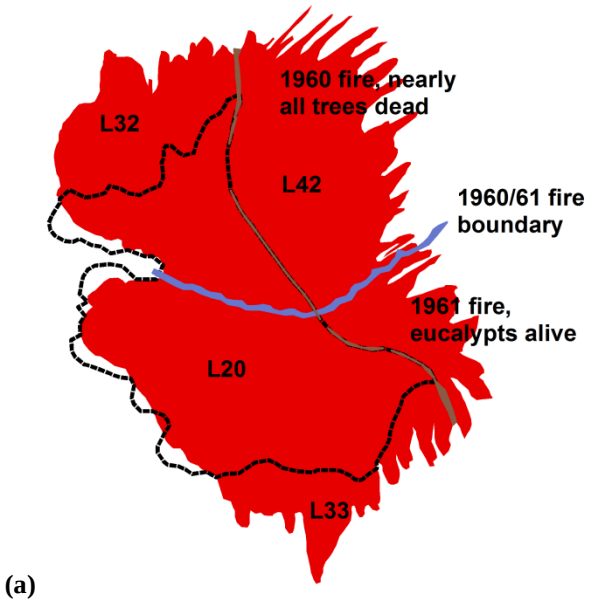
Increased carbon efflux (anthropogenic and non-anthropogenic) increases climate heating— the greenhouse effect. Increased carbon efflux increases vegetation mass through CO_2 fertilisation, e.g. the early phase of climate change which increased woody biomass.

11.4 Most pressing needs for further research

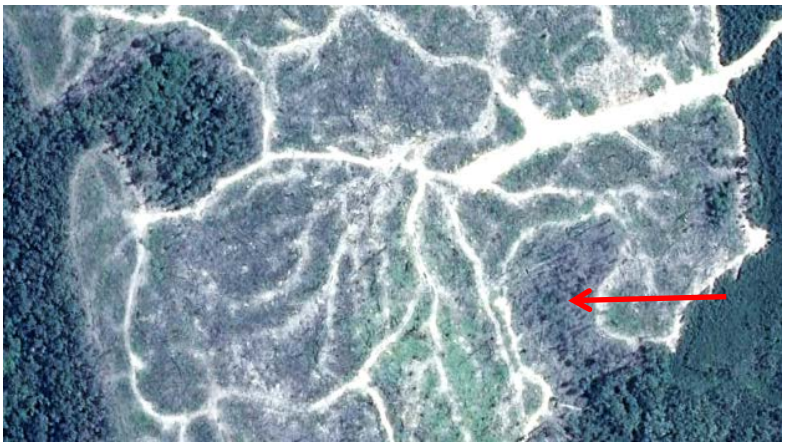
The project improved knowledge of C emissions, large trees, rangeland degradation, soil carbon, aspects of ecology and useful methodologies. However the situation that a comprehensive carbon budget for LUC requires more data (e.g. for forestry, Bradshaw et al., 2013), still remains. The project highlighted specific questions and the opportunities to answer them are discussed below.

The 3D modelling of tree shape and their surroundings showed a way to record and interconnect detail of their allometrics and autoecology such as SOC concentration derived from the individual trees. That part of the project was instigated in part as a foundation for when measuring SOC either side of the boundary of a logging coupe, in order to find the nearly instantaneous emissions specific to the logging activity, i.e. before the soil was influenced by post-logging differences in biomass and biomass decomposition. The unlogged side would have had to have an ‘escaped regeneration burn’, so that the only differences across the coupe boundary were the

mechanical part and the burn intensity of the logging. Both sides would have large trees, although on the logging side their roots and surrounding soil may sometimes



(a)



(b)

Figure 11-2. Escaped regeneration burn examples for SOC change analysis

(a) Coupe L20, Lords Block, Florentine Valley, Tasmania, 12.14 ha, logged in ~1960. Redrawn from the ANM coupe registers, access courtesy of Leigh Edwards, Forestry Tasmania, 2012. Escaped burn in the north across to east side of road killed most trees and therefore a good sample for comparing SOC change with north part of L20, whereas southern escaped burn did not restart eucalypt growth from scratch— not a valid comparison. **(b)** GoogleEarth© image, 2015, burnt vestige of forest as aggregate in lower right of coupe, Tarkine (Lat/Long -41.156403/144.844978): if it regenerates similar to neighbouring clearfell zone then it will proved a good sample for comparing SOC change.

be highly disturbed. On the unlogged side it would not be possible to measure SOC directly underneath or nearby the trunks (without damaging trees— hence the work undertaken here) whereas that SOC on the logged side may be spread around. Recently, only ~30 ha of forest are subject to escaped burns per annum (Scott et al., 2012). But that allows scope for measurements at different durations since logging. The escaped burn could be from the coupe-wide clearfell into neighbouring forest (CBS logging, Figure 11-2.a) or from the clearfell zone into the vestigial retention islands that are part of 'aggregate retention' logging (ARN logging, Figure 11-2.b) (0.9 ha and 2.6 ha per coupe respectively, Scott et al. (2012)).

Although the allometrics for large *E. regnans* has recently been improved (Dean et al., 2003; Dean et al., 2004; Dean and Roxburgh, 2006; Sillett et al., 2015; Chapter 5), the high-end allometrics still require more data, to improve the accuracy of carbon accounting of such logging, both past and future. The allometrics must cover different environments. The much-studied *E. regnans*–dominated primary forest in the Victorian Central Highlands (Galbraith, 1937; Ashton, 1975a; Lindenmayer et al., 1999; Ashton, 2002; Mackey et al., 2002; Keith et al., 2009), are under 300 hundred years old (Ashton, 2002) and the latest allometric study examined their specimens without senescence (Sillett et al., 2015) whereas the large Tasmanian ones are over 500 years old (Wood et al., 2010), generally have larger DBH, have varying degrees of senescence, have endured quite different fire histories, and have different understories and large-hemi-epiphyte ecology. The carbon accounting for these two populations under LUC, including that of their soil, and under climate change, should be differentiated. *E. regnans* in NE Tasmania, in the Otway Ranges in Victoria, and in SE Victoria, possibly represent further variation.

The information on some of the main pathways of carbon flux are numerically indeterminate and some of the processes are not well understood. For example, how much carbon descends to what depth in highly-fractured bedrock (the C horizon), how far is it transported horizontally by groundwater, and what are the emissions along the route? Relatively simple quantities such as the decline in annual carbon increment of grazed [native] cypress trees or various thickening species are not

presently known at a level applicable to either current carbon trading schemes or to include in LUC or climate modelling.

The modelling in this project did not include any change in growth rate due to climate change when forecasting the carbon balance of primary-forest logging or rangeland re-sequestration. More detailed knowledge is required on the [non-linear] growth rates for forests and reforestation, including under climate change conditions. A high SOC concentration underneath large trees' trunks in the mixed-forests was found but what happens to this if more fires (accompanying climate change) burn the tree hollows more frequently? This carbon pool was diminished even after one fire in the Styx Valley, even if the trees remained alive, and on cursory visual inspection the under-trunk concentrated carbon pool was not apparent in the more-burnt, dryer forests of Australia.

From observation of the literature it cannot be stated too often that due to the longer limiting half-life of SOC than that of biomass, experiments for SOC change analysis require long time spans. This must be considered in experimentation. More generally, definitive experiments would probably be more successful if examining large-area contrasts and with adequately paired sites (e.g. in the rangelands— pronounced piospheres or long-term, long-distance, intense fence-line contrasts; in the production-forests— escaped burns neighbouring logged areas). Monitoring requires establishment of large control regions, because large areas allow inclusion of spatial heterogeneity and limit contamination by edge effects.

Regarding LUC of the primary tall open-forests, for comprehensive carbon accounting a large effort is required to follow the path of the timber that has long since been removed from those forests and whose residues are now distributed across Australia and overseas. Similarly for eroded SOC in rangeland: measurement of that SOC flux, its transport and deposition, and its overall half-life, represent the most difficult empirical tasks for assessing SOC flux accompanying the LUC to grazed rangeland.

It would be difficult at the present stage, though not impossible, for any project in Australia to amply measure the changes in all major pools (soil, woody vegetation, mobilised soil carbon, and necromass) for the LUCs of primary forest conversion, land to rangeland grazing, or destocking rangeland, to any degree sufficient for carbon trading.

The estimates of national-level C emissions for the Australian rangelands have low precision, reflecting the state of knowledge of the historical and present C stocks. A relatively small portion of the rangelands has been sampled for C content, and an even smaller portion measured for change in C. As of 2001 only 44 sites existed in the rangelands that met the requirements of peer review, mature growth and metadata necessary for modelling potential biomass stocks (Barrett et al., 2001) and the precision of these biomass estimates was no better than ~25% of the mean. For biomass alone the task is very large, with dominant vegetation types ranging from Bottle Trees (*Brachychiton rupestris*) to Brigalow (*Acacia harpophylla*), Old Man Saltbush (*Atriplex nummularia*), Desert Sheoak (*Allocasuarina decaisneana*), Ghost Gum (*Corymbia aparrerinja*) and Spinifex grass (*Triodia* spp.). Total (aboveground and below ground) biomass and its growth rate must be determined, including under the changing climate, grazing and fire conditions. Grouped biomass change, at the ecosystem-level as in the NSW study (Chapter 10), may be more tractable but the remotely-sensed data still requires ground-truthing. Effort must be placed into finding quality benchmarks, either intact ecosystems or if only their most-intact vestiges, then estimates of how far those carbon stocks are below potential. Values derived from the benchmarks must be adjusted to correspond to the future climates before using them to forecast sequestration on neighbouring degraded land.

From work undertaken in Western Australia's rangeland (Dean et al., 2009) the costs of sampling SOC representatively down to about 1 m depth is ~AUD$20,000 ha^{-1} at current prices. The equivalent analysis for mapping the C stocks and potential for Australian rangelands at a spatial resolution of 20 km and a quantitative precision of 25% (e.g. for assignment of a commercial carbon cost) across Australia's 380 Mha of commercially grazed rangeland would prohibitively cost billions of dollars.

Routine rangeland and timber-production-forest monitoring (for land-condition assessment of leasehold properties and timber inventory respectively), although undertaken in various forms in different States and Territories for decades (e.g. Stone, 1998; Watson et al., 2007; Lawley et al., 2016), does not typically yield data applicable to C accounting. The technology to do so however is available. Monitoring of Australian landscapes for carbon conservation and fluxes lags behind biodiversity monitoring and [commercial-] 'resource' monitoring.

The new, national-level Terrestrial Ecosystem Research Network (TERN) initiative offers a common monitoring methodology nation-wide for biomass and SOC and the possibility of access to data for scientists. If adopted thoroughly, with appropriate spatial resolution, if LUC history is recorded, and if existing data in their various forms are translated into its database, then TERN should be very useful for carbon accounting. TERN offers integration of carbon and biodiversity monitoring along with some other attributes of importance to humans, the so-called 'ecosystem services' (van Dijk et al., 2014; Lindenmayer et al., 2015) but for the carbon aspect it relies on minimal funding and cooperation above that already applied in the entrenched systems.

It is possible to envisage an era where in many places the extra heat of climate change influences people's daily life as much as does road traffic today. In such a situation all medium to large trees will be recorded digitally and visible online, as are nearly all streets presently visible online via GoogleEarth©. Before that era, knowledge on carbon accounting for both forest logging and rangeland commercial grazing could be greatly improved beyond the present work if there was much greater data transfer between industry, government departments and the scientific community. For example, if, when logging primary-forests details were recorded of all trees felled, allometrics could be much improved. Coupe weigh-bills could be stored digitally, the mass checked at all wood-processing stages, and the data made publicly available. For rangeland the equivalent process would involve releasing high-resolution land-condition, biomass and infrastructure mapping, property-level climate data and spatio-temporal [livestock] stocking rates and [livestock] turnoff

rates, and recording data on trees felled. These data need to be all available online so that undertaking of carbon accounting work is not restricted to those people who have ready access to a particular government facility. A mixture of online and personally supplied data was used in the present study. Future studies would greatly benefit from access to existing data withheld from public access.

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Appendix I. Glossary

$^{-1}$: per. Used for units where a rate or concentration is expressed. E.g. Mg ha^{-1} (mass per unit area), or dse ha^{-1} $year^{-1}$ (number of dry sheep equivalents per unit area per unit time).

buttress region [of a tree]: for mature *E. regnans* it is anatomically complex. It starts where the lateral roots merge with the soil, which can be below 0 m on the downhill side of the tree, and extends up to 9–18 m up the trunk where the bark changes from sub-fibrous to the smooth ('gum') bark (Ashton, 1975a). 'Spurs' are the ridges of the buttress which join the main trunk to the large lateral roots. 'Flutes' are the valleys between spurs (Julin et al., 1998).

CO_2: carbon dioxide, the substance formed when carbon is oxidised (by burning or respiration, or the very final end product of decomposition of organic carbon). It is the substance that the general public often assumes is referred to when the media or politicians mention carbon emissions. This misconception arises in part because of the use of the CO_2-e (carbon dioxide equivalent) which is a common denominator used to add up the effects from a variety of greenhouse gasses upon the climate. To convert from an amount of C emissions to an amount of CO_2 emissions (or CO_2-e emissions), simply multiply by the ratio of their atomic weights: 3.6641. (And to convert from a mass of CO_2 (or CO_2-e) to a mass of C, divide by 3.6641.)

coupe: Forestry operations in Tasmania are delineated by logging coupes— each with a certified plan, covering ~70 ha (Wilkinson, 2003; McIntosh and Ware, 2008).

DBH: tree diameter at 'breast height', i.e. at 1.3 m above the solum surface, in Australia (measured at different heights in different countries). Usually determined by measuring the best-fit circumference and dividing by π.

flux: amount of substance moved, or a measure of field strength, per unit area per unit time. Typically used for e.g. to describe a magnetic field or a flow of water through a hole. With respect to carbon accounting this may typically be the amount

of carbon in tonnes per hectare per year, either being sequestered or emitted; or in derived SI units: Mg per ha per year.

footprint [of a tree]: the area inside a convex hull polygon that circumscribes where the large lateral roots merge with the forest floor.

forest floor: includes the litter, fermentation and humus layers above the mineral soil (above the solum).

humus mound: the pyramidal-shaped fermentation and humus material around the base of eucalypt trees. The humus mound excludes the litter layer, as on a volume basis it often consists of eucalypt branches which can be readily discounted during measurement. Humus mounds are called 'duff mounds' in the USA (Ryan and Frandsen, 1991).

hectare: unit of area, symbol ha, equal to a square of size 100 m by 100 m, or a rectangle 5 m by 2,000 m; etc.

lignomor: decomposed, structure-less wood. A term from the humus soil classification tuple of Green et al. (1993), Klinka (1997) and Englisch et al. (2005) is used to describe the macroscopic, low pH, humic soil, both aboveground and belowground, derived from wood (trunk, branches and coarse roots) without the original plant structures being macroscopically discernible.

Mg: megagramme, derived SI unit of mass. 1,000,000 grammes = 1 tonne (Mega meaning one thousand thousands = 10^6 = 1 million). (Note that Mg is also the chemical symbol for magnesium, but that isn't referred to in this thesis.)

Mha: derived SI unit of area. 1,000,000 hectares = 10,000 km^2 (Mega meaning one thousand thousands = 10^6).

myrtle: myrtle trees, *Nothofagus cunninghamii* (Hook. f.) Øerst., recently renamed *Lophozonia cunninghamii* (Hook. f.) (Heenan and Smissen, 2013), commonly known as myrtle beech or myrtle. Arguments have been made suggesting that the renaming need not be recognised (Hill et al., 2015).

oldgrowth forest: Forest stands that have reached a certain age or late successional stage and that may or may not have been impacted by humans (Wirth et al., 2009b). The word oldgrowth is often seen written in its earlier hyphenated form: 'old-growth', but as the forest fieldwork in the present work focuses on Tasmania I adopt the non-hyphenated form of Hickey (1994).

Pg: petagramme, derived SI unit of mass. 1,000,000,000,000,000 grammes = 1,000,000,000 tonnes = 1,000 Tg (Peta meaning one thousand, thousand, thousand, thousand, thousands = 10^{15} = 1 million million thousands).

primary forest: Forest stands that have been subject to very low levels of human impact for an extended period of time, including stands of any age and time since disturbance (Wirth et al., 2009b). The term is used in this thesis to differentiate primary forest from the secondary forest that is the product of, for example, logging, in an analogy to the forests affected by slash and burn agriculture in the tropics, and in an attempt to be comprehensible to an international readership. Most Tasmanian primary wet-eucalypt forest is 'oldgrowth' in the Tasmanian government's terminology by being largely dominated by trees more than 110 years old, but some is regrowth, dating from wildfires less than 110 years ago.

snags: standing dead trees, in the UK: stags (due to appearance resembling stags' horns)

SOC: soil organic carbon, usually in units of Mg ha^{-1}. It can refer to the entire soil profile or to just the measured part of the profile— the reader must be aware of the context.

Δ**SOC**: change in soil organic carbon, usually in units of Mg ha^{-1} $year^{-1}$, or Mg ha^{-1} with a time period described at length. As for SOC, ΔSOC can refer to the entire soil profile or to just the measured part of the profile.

Tg: teragramme, derived SI unit of mass. 1,000,000,000,000 grammes = 1,000,000 tonnes = 1 Mtonne (Tera meaning one thousand, thousand, thousand, thousands = 10^{12} = 1 million million).

turnoff: the number of commercial livestock exiting a rangeland property, either for sale or culled.

Appendix II. Myrtle coal (cunnite).

A biogenic, high-carbon, high-lignin stone with conchoidal fracture from myrtle beech (*Nothofagus cunninghamii* (Hook. f.) Oerst.) in Tasmanian mature primary forests

A-II.1 Abstract

A hard, moderately uniform, solid material that results from natural decomposition of the wood of mature *Nothofagus cunninghamii* (Hook. f.) Øerst. (myrtle beech) trees was discovered in Tasmania. The substance has been named cunnite, being derived from the original species name and suffixed with 'ite' to indicate that it is a rock. Cunnite has conchoidal fracture (indicating isotropic bonding), whereas all other tree decomposition products noted in the same forests were readily friable, with no evidence of intermolecular bonding. It is a dehydrated form of lignomor and its pre-dehydrated from also has conchoidal fracture. Cunnite was examined by carbon assay, radiocarbon dating, scanning electron microscopy and infra-red spectroscopy. The spectroscopy suggests that it is high in lignin, and therefore it is most likely formed by a brown-rot fungi. Although it appears to last beyond the decomposition of the entire tree, the age determined from radiocarbon dating does not suggest that it is long-lived compared with the tree age. A synthetic version of the material may be useful for hydrogen storage. Cunnite is part of the forest carbon cycle but forms a small fraction of the soil organic carbon pool.

A-II.2 Introduction

Conchoidal fracture in biogenic substances is rare, being found in: some animal spines and shells (Berman et al., 1993), grain husks, air-dried plant resins (Simmonds, 1895), anthracite (Stopes, 1919), and some high-lignin nut shells (Van

der Pyl, 1948). Conchoidal fracture is common however in lignin-based activated carbon synthesised from black liquor and pyrolised Kraft lignin, waste products from wood pulping (e.g. Sharma et al., 2004; Maldhure and Ekhe, 2011); and the high-lignin form of fossilised timber, vitrinite (Kaelin et al., 2006).

The commonest source of lignin— waste from wood pulping, including that from primary forests, has most often been burnt as fuel (Ragauskas et al., 2014), thereby increasing greenhouse gas concentrations relative to using low-emission renewables. The uses of lignin are increasingly diverse and its chemical and physical properties remain a vibrant topic of research (Norgren and Edlund, 2014). Understanding a form of lignin with new structural properties may help to elucidate the underlying chemical nature of lignin in general and thereby find a more environmentally friendly use for it. As a climate-change mitigation activity, natural lignocellulose materials with micropores are being investigated for hydrogen storage, to enable hydrogen fuel usage in place of fossil fuels, (Babel et al., 2012). They are also useful for metal pollutant absorption (Suhas et al., 2007) and potentially for absorbing dietary fats (Tolba et al., 2011). Natural high lignin-content materials such as walnut shells (Van der Pyl, 1948), with natural isotropic bonding (as evidence by conchoidal fracture) may represent a more durable material than that derived from black liquor from wood pulping. Determination of the chemical reasons for the differences between lignin types in nature should help the artificial synthesis of such material, as would be necessary for market demand.

The main carbon pools of forests and the main routes of movement of carbon from air to tree to soil and back to the atmosphere are well-established. The pools are living plants, animals and trees (biomass), coarse woody debris and fine litter, soil organic carbon, solum fauna and dissolved organic carbon. Uncertainty in anthropogenic emissions accompanying land use and land cover change is the largest unknown in the global carbon budget (Houghton et al., 2012). Soil organic carbon (SOC) emissions are the next most significant after biomass emissions, accompanying land use change (LUC); but relative to other pools, changes in SOC and roots are poorly known (Kim and Kirschbaum, 2014; Scharlemann et al., 2014).

Nevertheless the stocks and halflives of different soil carbon pools and some types of decomposing wood are still being elucidated, as are the spatio-temporal routes of some translocated, dissolved organic carbon and carbon near the bedrock. SOC occurs in many forms and often whether some is assigned to the carbon pool of dead wood or to soil carbon, is a grey area. It can depend on historical categorisation systems, visibility, and ease of measurement (e.g. Mobley et al., 2013). For example, decomposed roots underground have been called 'soil wood' (Jurgensen et al., 1997) (technically a lignomor) but are too soft to detect as one detects wood, whereas the equivalent stage of decomposed timber (also 'soil wood') aboveground or in the forest floor is classed as coarse woody debris, i.e. wood and not soil. Determining the nature of the intermediate phase between wood and soil should help refine our knowledge of the carbon cycle, for use in climate change forecasts dependent on land use.

Reported here are preliminary investigations into a recently discovered substance from inside the trunks of mature, rainforest hardwood trees of the species myrtle/myrtle beech (*Nothofagus cunninghamii* (Hook. f.) Øerst.). It is a timber decomposition product (lignomor) with conchoidal fracture, which appears chemically and physically stable when dry and in the form of a stone. It was found in the Styx Valley, in the Australian State of Tasmania, but could also occur in myrtle trees elsewhere in Tasmania or in the State of Victoria.

Insights into the chemical composition of wood species and into the change in chemistry during decomposition of wood have been provided by Fourier transform infra-red spectroscopy (FT-IR) (e.g. Faix, 1991; Facklera et al., 2010; Yilgor et al., 2013). FT-IR is deployed here to help elucidate the origin of myrtle coal. Other techniques deployed were scanning electron microscopy, radiocarbon dating, and carbon assay. The aim of this brief investigation was to determine, the origin and nature of the substance, what insight it may provide to forest processes, and if a synthetic version could be formed, for use in chemical processing or energy storage as discussed above.

A-II.3 Methods

A-II.3.1 Study area and sample collection

The samples were obtained from several sites in mixed-forest in the Styx Valley (Figures 5-2, 6-1) over an elevation range of 300–670 m.

The first sample of cunnite was found in a myrtle tree that had been pushed over during logging, DBH 1.76 m, height 32(4) m (section 5.3.5.3, Figure 5-9, Figure A-II-1.b). Its roots were being examined for a soil carbon experiment and in the trunk hollow several kilograms of dried decomposition product were found. In the centre of this product was the hard rock form, with visual similarities to coal (specimen CD089). It was initially named myrtle coal, then cunnite, to represent its species name and indicate it was a rock and a phase in biogenic rock formation. Technically it is a soil of the type lignomor.

The second sample was the wet lignomor, possibly still undergoing decomposition, although no timber structures were visible (specimen CD079). Both substances had conchoidal fracture. This wet material was called ‘myrtle cheese’ owing to its texture being very similar to cheese, e.g. Edam, Jarlsberg or Nimbin (rubbery, soft cheese). The myrtle cheese was in a myrtle that had been pushed over during logging. The fact that both the myrtle cheese and the cunnite were only found in myrtle trees, and that both had conchoidal fracture, led to the assumption that the cunnite was formed from myrtle cheese after dehydration, which was checked by comparison of their spectra.

No material with structure similar to cunnite or myrtle cheese was found in any other tree species. It was not uncommon however for mature eucalypts in mixed-forest to form lignomor in their hollow centres, noted after logging; none however had lignomor with conchoidal fracture, i.e. any significant bonding between the molecules.

Not all decomposing myrtle trees had the myrtle cheese or cunnite. Two myrtle trees were found with a non-conchoidal lignomor which was light and friable and appeared to have no bonding between its particles (specimens CD064 and CD065).

Table A-II-1. Myrtle wood and lignomor specimens.

type	sub-type	speci-men #	latitude/ longitude (°)	elev-ation (m)	position	tree DBH (m)
myrtle wood	wood	CD091	-42.820667/ 146.612167	320	1.3 m above solum	1.76
myrtle coal, cunnite	dried, conchoidal	CD089	-42.820667/ 146.612167	320	0.30 m above solum, inside pushed-over tree	1.76
myrtle coal, cunnite	dried, conchoidal	CD103	-42.814500/ 146.549167	500	forest floor	-
myrtle coal	dried, partly conchoidal & massive	CD121	-42.823333/ 146.613000	391	0.1 m above solum, inside decomposing, dead tree	<1
myrtle cheese	wet	CD079	-42.795333/ 146.588667	651	1.3 m above solum, inside pushed-over tree	~1.1
non-conchoidal, decomposed myrtle trunk	friable, damp	CD065	-42.795545/ 146.585867	612	0.335 m above solum, inside cut stump	~0.8 m
non-conchoidal, decomposed myrtle root	friable, damp	CD064	-42.795663/ 146.588496	640	-0.3m, hollow small myrtle	~0.6 m

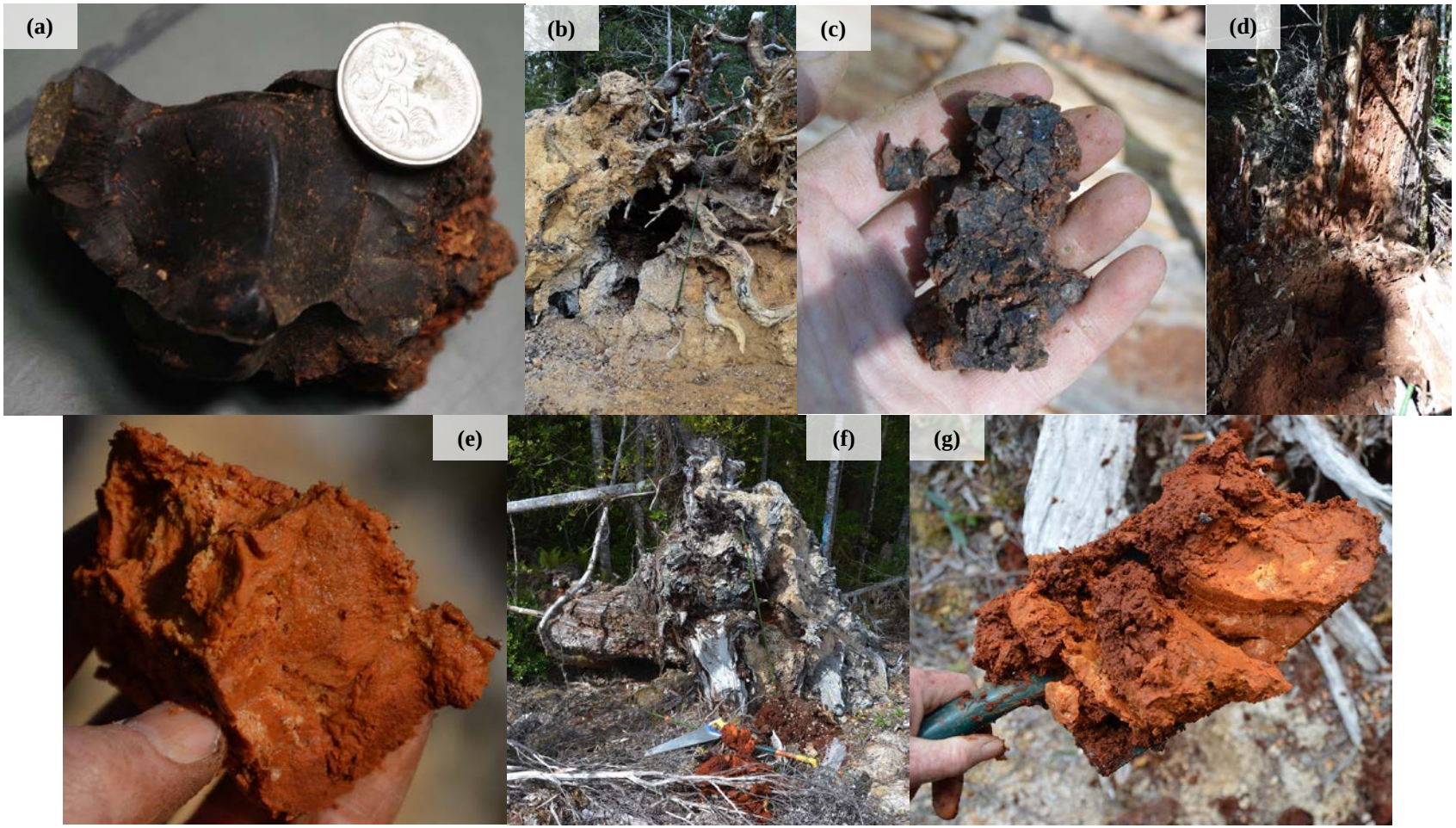


Figure A-II-1. Lignomor location and samples. **(a)** myrtle coal (cunnite), portion of CD089, coin diameter= 19.41 mm, **(b)** location of CD089, **(c)** CD121, **(d)** location of CD121, **(e)** CD079, **(f)** location of CD079 **(g)** surrounds of CD079, showing two textures and colours.

Where space was limited, inside the confined tree trunk, an Eijkelkamp© soil corer (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands) was used to sample soil for carbon and bulk density following the procedure described in Roxburgh et al. (2006a). For the wet myrtle cheese (specimen CD079) the hand-held bulk-density head from the Eijkelkamp kit was used.

A-II.3.2 Sample processing and analysis

The sample processing procedure for carbon assay and pH is described in section 6.3.3.

To help confirm the origin of some of the charcoal and the two samples CD089 and CD103 they were carbon dated by James McDonald and Paula Reimer of the Chrono Centre, Queens University, Belfast, Northern Ireland. The charcoal was from the central root zone below a myrtle that was examined for root architecture and it had the shape of a young twig: it therefore had probably been grown just prior to the fire and therefore its radiocarbon date was similar to the fire date. Both the charcoal and cunnite specimen CD089 responded to routine 'AAA' processing: 4% HCl @80°C 1-2 hours, 0.5% NaOH @20°C 1 hour, 4% HCl @80°C 1 hour, rinse with deionised water until neutral then dry. Cunnite specimen CD103 did not respond to routine sample extraction and its pre-treatment was tailored: 'humics' extracted, 4%HCl @ 80°C 2 hours, H_2O wash, 0.5%NaOH @50°C 2 hours, acidified 4%HCl, centrifuged, liquid decanted and discarded, 'humics' freeze dried.

One sub-sample of cunnite specimen CD089 was also analysed by scanning electron microscopy, led by Karsten Goemann at the Central Science Laboratory, University of Tasmania. The SEM sample was carbon coated.

A-II.3.3 FT-IR spectral analysis

Samples of myrtle at different stages of decomposition and along different routes of decomposition were analysed by FT-IR spectroscopy. To find why myrtle decomposed differently to other trees samples of the following species were also analysed: swamp gum (*Eucalyptus regnans*), celery-top (*Phyllocladus aspleniifolius*

Labill.) and sassafras (*Atherosperma moschatum* Labill.). The spectral runs were performed by Thomas Rodemann on a Bruker Vertex 70 infrared spectrometer, at the Central Science Laboratory, University of Tasmania. Automated matching of cunnite spectra against a spectral library was performed by Thomas Rodemann.

To allow comparison between the FT-IR spectra for the specimens, separate spectral graphs were produced with the spectra normalised at two absorption minima: 1176 and 1485 cm^{-1}, three absorption peaks: 1109, 1034, and 1597 cm^{-1}, before the water peak range at 1670 cm^{-1}, and at the low wavenumber edge of the water peaks— 2400 cm^{-1}. Spectral features recorded were peaks and absorption differences (i.e. increases, decreases, and peak shifts). Spectra of the decomposition samples were compared with the myrtle wood in regards to change in relative peak heights and the formation of new peaks, and were assigned qualitative entries (e.g. absent, decrease, increase, shift) with respect to the wood specimen. The majority of the spectral interpretation was performed by Ernst Horn, of the Department of Chemistry at Rikkyo University, Tokyo, Japan.

A-II.4 Results

A-II.4.1 Basic characteristics

Bulk density, carbon density and pH are shown in Table 6-1 and Figure 6.7. The carbon wt% for cunnite at 52.3% was the highest of all soil and plant materials tested. The carbon wt% for myrtle cheese at 49.1% was lower than for charcoal but higher than for the other soil and plant samples (other than for cunnite). The carbon density for cunnite (804 Kg m^{-3}) is similar to that for anthracite. In Figure 6.7 cunnite is the outlier to the curve of best fit, to the right-hand-side. The hardness of cunnite was 4–5 on Mohs's scale; determined by Isabella von Lichtan in Geology, University of Tasmania. The conchoidal fracture suggested a form of hydrogen bonding that was absent in lignomor derived from eucalypts.

Surrounding pure cunnite was a lower grade material, with many air pockets, it was a mixture of brown and black, more friable material. The different grades due visibly to more air pockets (i.e. lower bulk density and C wt%), suggest that the curve in Figure 6.7 should be J-shaped (a minima and two bends) rather than hyperbolic with only bend.

It appears that upon long-term drying the more uniform lighter coloured material (like specimen CD079) becomes cunnite (like specimens CD089 or CD103) and that the darker material with air pockets becomes the lower grade cunnite (like specimen CD121).

A-II.4.2 Radiocarbon dating

The radiocarbon dates of the cunnite specimens, CD089 and CD103 were 1811(±22) and 1730(±22) respectively (ages 202 and 283 years when found in 2013). I.e. the one found on the forest floor was ~80 years older, but they are both well within expected longevities of myrtle trees so no statement can be made about the halflife of the cunnite itself. If the samples were from timber that formed when the tree was quite young then the age determined is that of the trees (which is possible as the substance in its purest form is found just above ground level in the centre of the trunk). Specimen CD089, being inside the trunk hollow, aboveground and with the tree vertical and alive prior to logging, would remain dry and not receive UV light, and therefore would probably remain intact at least until the tree had died and decomposed, which could have been another 100 years. After which it would be on the forest floor as was CD103. The durability of the cunnite in the open is unknown.

This gives some indication of the longevity of part of the myrtle trunk, a region within the hollow that did not form a carbon flux to the atmosphere. The twig charcoal from in the root zone of the same myrtle that contained CD089, had the radiocarbon date of 242(±22) BC, indicating a forest fire~ 2260 years prior to 2015.

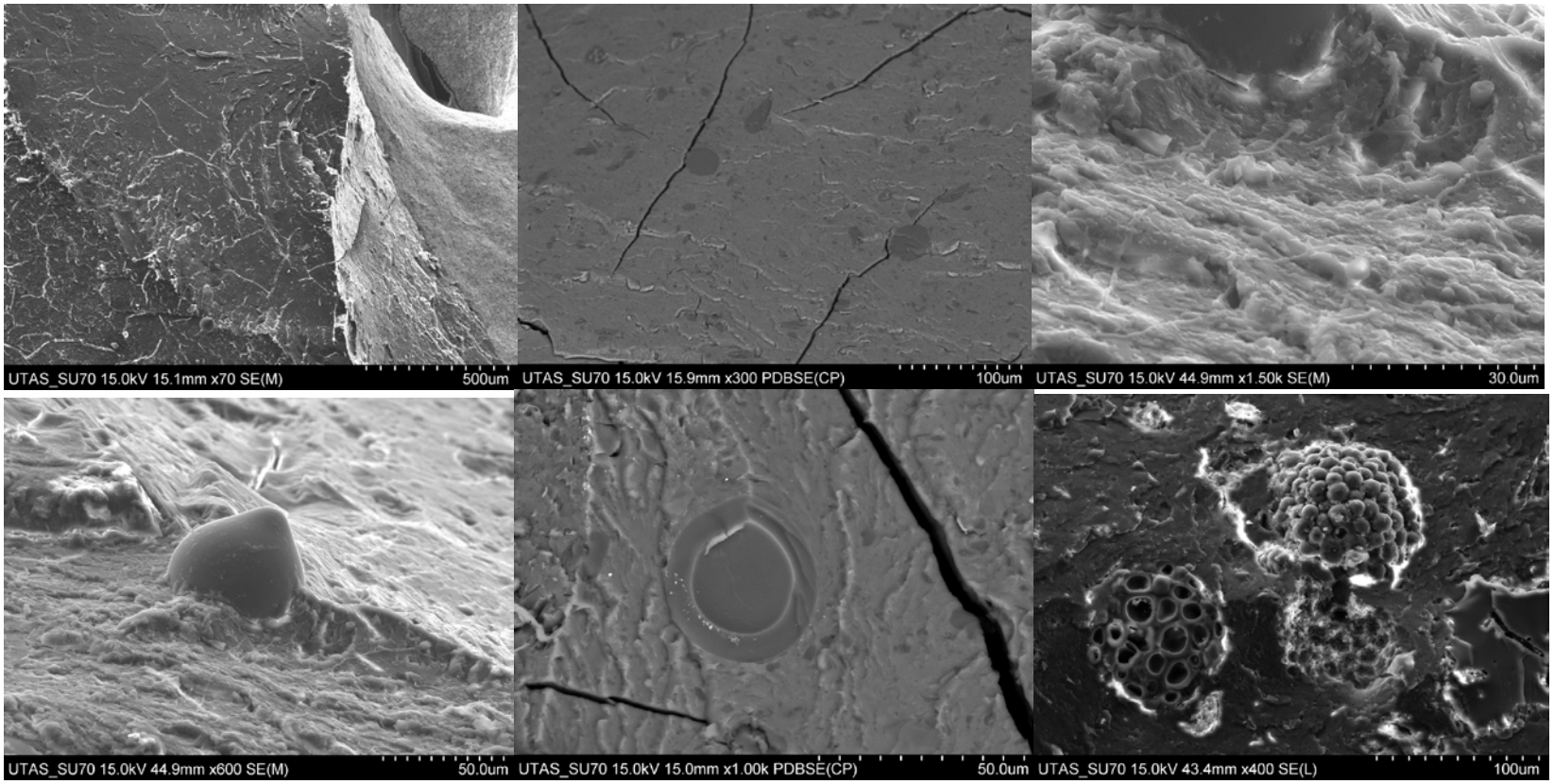


Figure A-II-2. SEM micrographs of cunnite specimen CD089 showing conchoidal fracture to microscopic-level and some inhomogeneity, embedded clusters of capsules, and single capsules.

A-II.4.3 SEM

The scanning electron microscope (Fig. A-II-2) revealed that the conchoidal fracture persisted down to at least the 10 μm scale. There was inhomogeneity in the form of inclusions (Fig. A-II-2.b), clusters of capsules, and single capsules. The capsules were 45±5 μm wide and some were either broken or had opened. Strands less than 1 μm wide, were occasionally evident (Fig. A-II-2.b). They may have been fungal hyphae though hyphae are typically an order of magnitude wider. It could not be determined whether or not the capsules were from the fungi that had caused the decomposition or had fallen down from some organism higher up in the tree hollow.

A-II.4.4 Infrared Spectra Analysis

The strongest match for cunnite against the spectral library was Kraft lignin (from paper making). Most other strong matches were different forms of lignin.

The spectra of the myrtle wood and its decomposition products (Table A-II-2) in the Infrared absorption range between 900 and 3000 cm^{-1}, were comparable with those in the decomposition sequence of *Liquidambar orientalis* Mill. (sweetgum) (Yilgor et al., 2013), and with general wood spectra and changes with for decomposition of Spruce (*Picea abies* (L.) Karst.) (Facklera et al., 2010).

The spectra of cunnite and myrtle cheese appeared very similar in the 'fingerprint' region, especially from 1000 to 1100 cm^{-1}, compared with the spectra of wood and the friable decomposed wood. Together with their similar origin and texture, this suggested that cunnite had formed from dehydration of myrtle cheese, i.e. that myrtle cheese was the precursor of cunnite.

The undecomposed wood samples, *Liquidambar orientalis* and myrtle wood (specimen CD091) can be compared, using the peak assignments in Table A-II-1. Apart from some weak- to medium-sized absorptions not seen or not assigned in the spectrum reported by Yilgor et al. (2013), the spectrum of myrtle wood is consistent with their results, within experimental error. Hence, the two wood types have the

same basic composition, namely both contain cellulose, hemicellulose, lignin and xylan molecular structures, as one would expect. The relative quantities of those structures are not compared in the present work. The extra peaks observed that are either not assigned or not observed by Yilgor et al. (2013) in the range 900-1800 cm^{-1} are at 1539 and 1612 cm^{-1}. The former is consistent with another aromatic-type vibration due to the benzene rings in one or more of the cellulose, hemicellulose, lignin and xylan wood components. The absorption at 1612 cm^{-1} is assigned to bonded (adsorped) water (Nakamoto, 1978; Horn and Snow, 1984; Borrega, 2011) or aromatic skeletal vibration breathing with C=O stretching (Kubo and Kadla, 2005; Facklera et al., 2010). In the absorption range 3000-3700 cm^{-1} the broad peak at 3370 cm^{-1} is due to ν(OH) bond absorption of several different R-OH functional groups H-bonding to atoms with different donating properties (Sharma et al., 2004);. In addition, superimposed in this area are ν(OH) absorptions due to absorbed water H-bonding to wood component functional groups (Nakamoto, 1978; Horn and Snow, 1984; Kubo and Kadla, 2005). The myrtle wood specimen has two extra peaks, 2852 and 2924 cm^{-1}, due to either bonded water (Nakamoto, 1978; Horn and Snow, 1984; Borrega, 2011) or C-H stretching (Kubo and Kadla, 2005; Facklera et al., 2010).

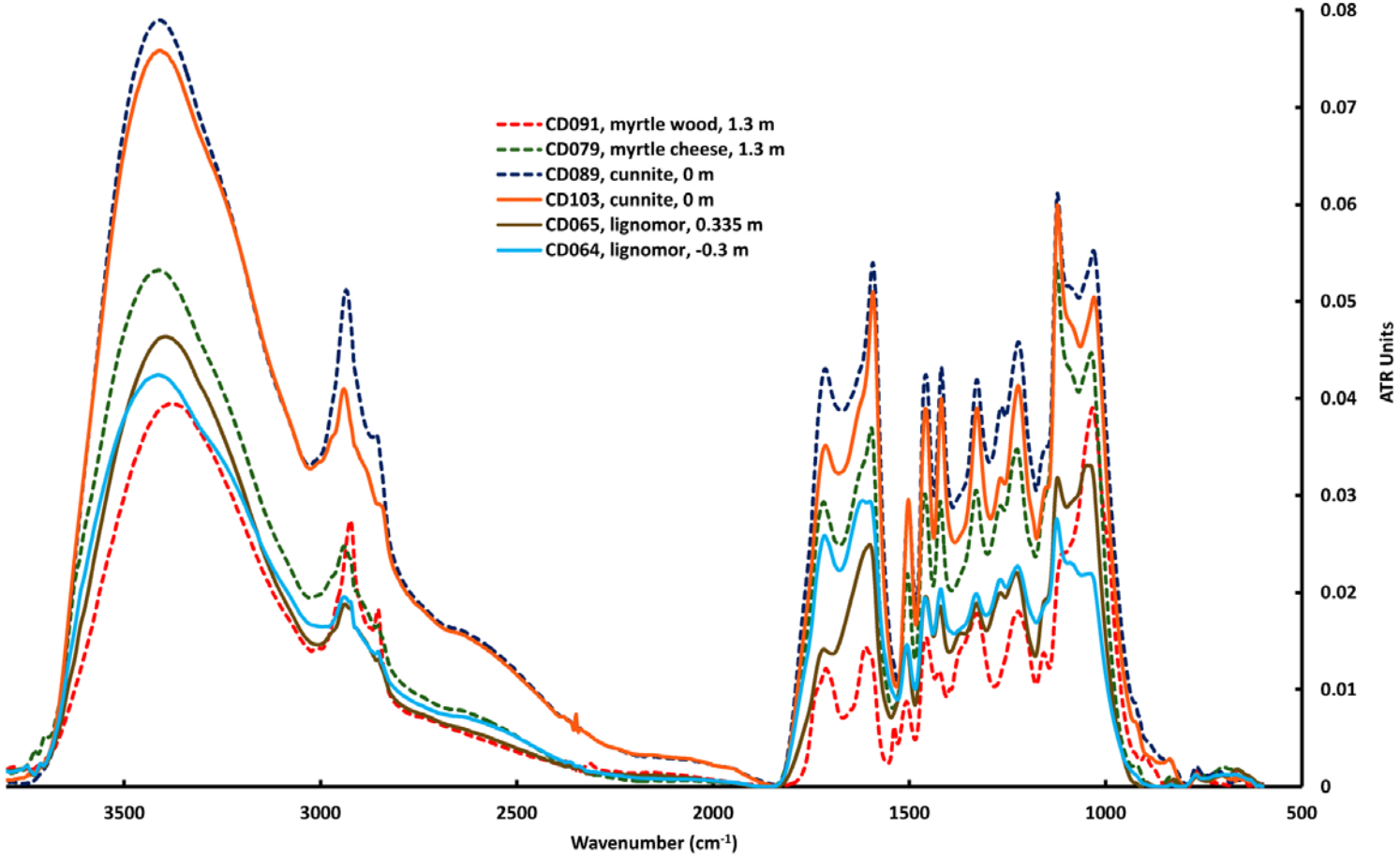
The decomposition samples CD079 (myrtle cheese), CD089 (cunnite), and CD103 (cunnite) can be compared with the myrtle wood. The new peaks in the range 900-1800 cm^{-1}, with the cunnite decomposition pathway are at (from Table A-II-2): 1122-1124 , 1150-1151 , 1265-1270 , 1595 , and 1630 cm^{-1}. Due to the close proximity to the corresponding undecomposed wood absorptions at 1157 cm^{-1} and 1093 cm^{-1} in the wood, peaks ~1123 cm^{-1} and ~1150 cm^{-1} above are consistent with at least a partial decomposition of the cellulose and hemicellulose. The peak ~1267 cm^{-1} is consistent with the decomposition of lignin and xylan due to the peak shift. The peak decrease at 1593-1601 cm^{-1} is consistent with the partial decomposition of the aromatic benzene rings of cellulose, hemicellulose and lignin. While the peak at 1612 cm^{-1} appears to have decreased, shifted and broadened, becoming weak shoulder peaks towards 1630 cm^{-1}, due to a decease and change in either bonded water or aromatic carbonyls. The general absence and decrease of the other absorptions shown in Table A-II-1 of the decomposition samples is also indicative of

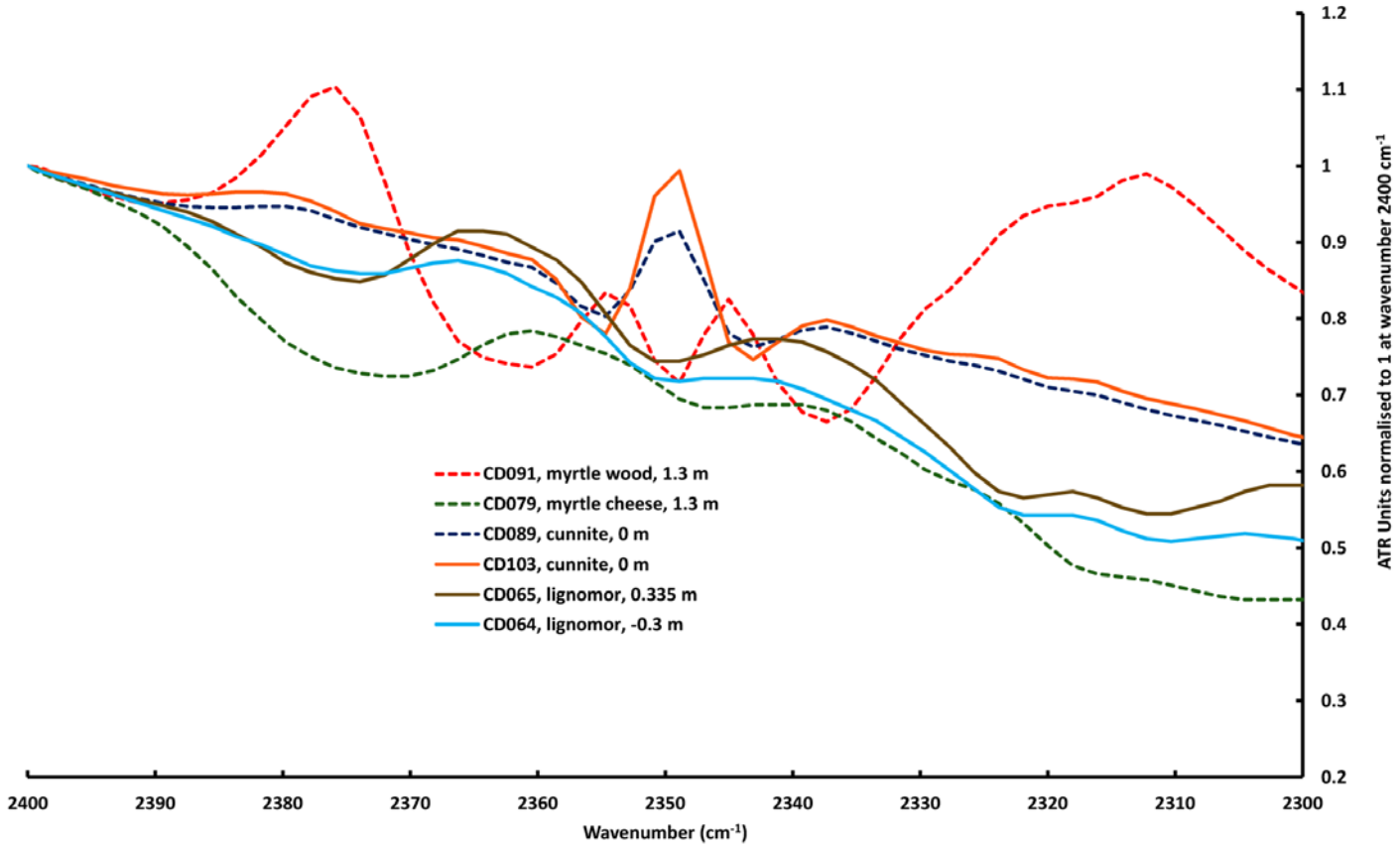
decomposition of the main wood components. The assignment ‘water loss’ refers to the decay sample containing less water, while ‘increase’ and ‘present’ refer to peak sizes that are larger than expected due to two or more superimposed absorptions.

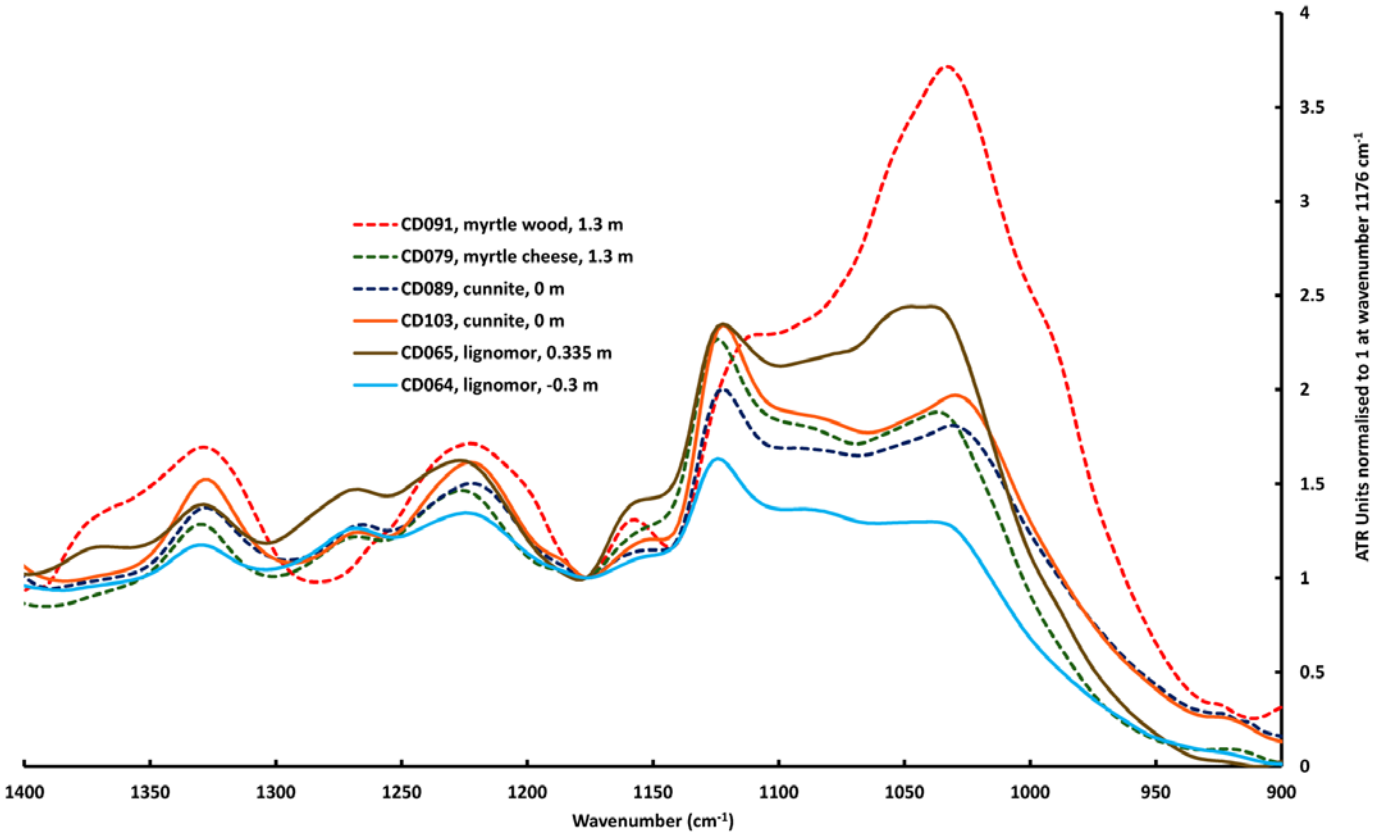
At 2349(±40) cm^{-1} there are several peaks, possibly being different shifts for the different specimens but cunnite appears to have a sharper single peak than the other specimens. These peaks are unassigned but may probably be related to hydrogen bonding of water, or ν(C-H) in the cellulose, hemicellulose and lignin.

The myrtle lignomors that showed no conchoidal fracture (CD064, CD065) had less-defined spectra with broadened peaks in most regions, e.g. near 1610 and 2900 $cm^{-1.}$

In comparison with other species the spectrum of myrtle has characteristics of both the hardwood, swamp gum, and the softwoods celery-top and sassafras (Figure A-II-3). Myrtle shows only a very weak shoulder peak at 1261–1265 cm^{-1} whereas the other species (both hardwood and softwood) show a definitive peak. At 1327 cm^{-1} it shows the strong peak characteristic of hardwoods (ν C–H in cellulose and ν C1–O of syringyl derivatives, Emandi et al. (2011)). Similarly, in the 1500–1800 cm^{-1} region it more-closely matched swamp gum, however it shows a weak shoulder at 1657 cm^{-1} that is absent from swamp gum but present as a peak for celery-top and myrtle. That region is characteristic of the water holding ability of wood species (Emandi et al., 2011). At 2349 cm^{-1} it showed the same lack of absorption as celery-top and sassafras, compared with the absorption by swamp gum. At 883 cm^{-1} and 924 cm^{-1} myrtle showed weak shoulder peaks, whereas swamp gum showed strong peaks and celery-top and sassafras showed no peaks. These results concur with myrtle being a hardwood timber but suggest that it may have a more diverse chemistry than either the hardwood or softwoods in the same forest.


CD091, myrtle wood, 1.3 m
CD079, myrtle cheese, 1.3 m
CD089, cunnite, 0 m
CD103, cunnite, 0 m
CD065, lignomor, 0.335 m
CD064, lignomor, -0.3 m
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
ATR Units
3500
3000
2500
2000
1500
1000
500
Wavenumber (cm-1)


CD091, myrtle wood, 1.3 m
CD079, myrtle cheese, 1.3 m
CD089, cunnite, 0 m
CD103, cunnite, 0 m
CD065, lignomor, 0.335 m
CD064, lignomor, -0.3 m
2400
2390
2380
2370
2360
2350
2340
2330
2320
2310
2300
Wavenumber (cm-1)
1.2
1.1
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
ATR Units normalised to 1 at wavenumber 2400 cm-1


CD091, myrtle wood, 1.3 m
CD079, myrtle cheese, 1.3 m
CD089, cunnite, 0 m
CD103, cunnite, 0 m
CD065, lignomor, 0.335 m
CD064, lignomor, -0.3 m
1400
1350
1300
1250
1200
1150
1100
1050
1000
950
900
Wavenumber (cm-1)
4
3.5
3
2.5
2
1.5
1
0.5
0
ATR Units normalised to 1 at wavenumber 1176 cm-1

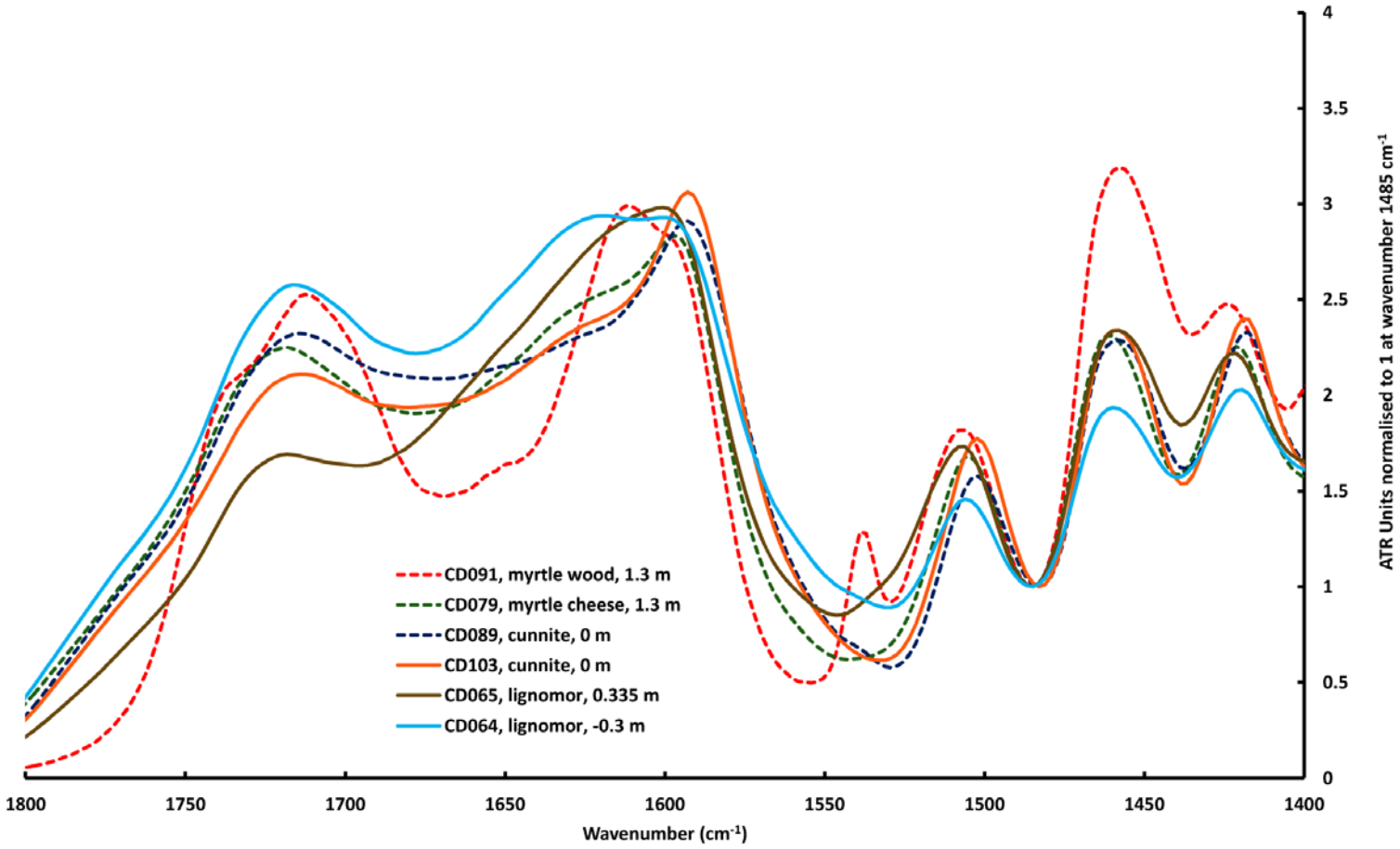


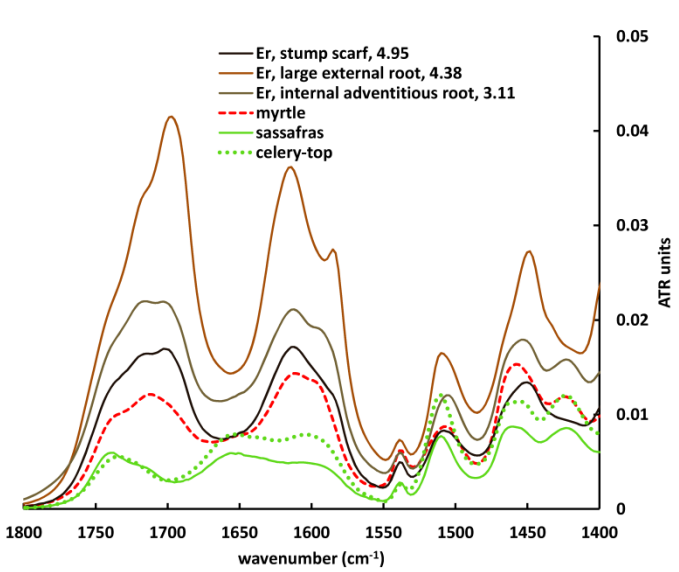
Figure A-II-2. FT-IR spectra. **(a)** entire spectrum in absolute ATR (attenuated total reflectance) units. (b-d) have been normalised.

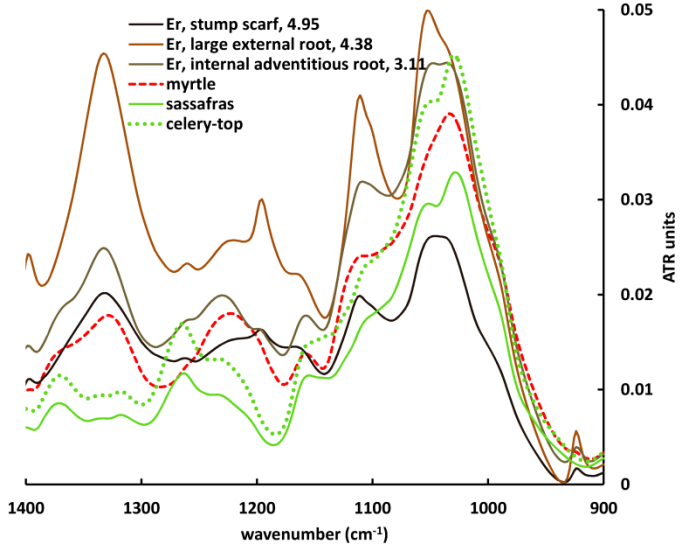
Table A-II-2. FT-IR spectral peak tentative assignments. Symbols for bond vibration: 'ν' stretch, 'δ' in-plane scissoring. Chemical abbreviations: 'car' carbohydrates, 'cel' cellulose, 'lig' lignin, 'pol' polysaccharides, 'xyl' xylan. Peak strength abbreviations: 's' strong, 'm' medium, 'w' weak, 'vw' very weak, 'vs' very strong. Peak shape abbreviations: 'br' broad, 'sp' sharp, 'sh' shoulder . Literature: [1] Yilgor et al. (2013), [2] Nakamoto (1978), [3] Horn and Snow (1984), [4] Fackler et al. (2010), [5] Kubo and Kadla (2005).

wood (CD091)			decomposition specimens				
absorption (cm^{-1})	strength, shape	assignments, wood	specimen	absorption (cm^{-1})	strength, shape	change	assignments
3370	s, br	ν(O-H) hydroxy groups [1,5], ν(O-H) water hydrogen bonding [2,3,4,5], or alcohol cel [4]	all	3379-3412	s, br	shift	water
2924	s, sp	ν(O-H) bonded water [2,3], or ν(C-H) alkane stretch cel, hem, lig [1,4]	all	2935-2941	br	decrease	loss
2852	w, sp	ν(O-H) bonded water [2,3], or ν(C-H) alkane stretch cel, hem, lig [1]	all	2840-2858		decrease	loss
2376	vw	ν(C-H) cel, hem, lig and xyl [1]	CD089, CD103	2380	w, sh	decrease	cel, hem, lig, xyl decomp.
2365	vw	ν(C-H) cel, hem, lig and xyl [1]	all	2360-2366		decrease	cel, hem, lig, xyl decomp.
absent			CD089, CD103	2349	w, sp	increase	unassigned
2312	vw	ν(C-H) cel, hem, lig and xyl [1]	all			absent	cel, hem, lig, xyl decomp.

1712	m	ν(C=O) xyl [1], ν(C=O) unconjugated ketones, carbonyls and ester groups [4]	all	1712-1718		decrease	xyl decomp.
1651	vw, sh	δ(H-O-H) water [1,2,3], and/or ν(C=O) xyl [1], ν(C=O) conjugated p-substituted aryl ketones lig [4]	absent				
1612	m, br	δ(H-O-H) bonded water [2,3], aromatic vib.; ν(C=O) [1]	all	1617-1631	w, sh	decrease, shift	
1597	m, sh	aromatic vib.; ν(C=O) [1], aromatic skeletal vibrations plus ν(C=O) lig [4]	all	1593-1601	m	increase	aromatic vib., ν(C=O) [1]
1539	w	aromatic vibration cel, hem, lig and xyl	absent				
1507	m	aromatic vibration lig [1, 4]	all	1502-1506			
1458	m	δ(H-C-H) lig, xyl [1]	all	1456-1460			
1423	w	aromatic vib. and (C-H) def.; and (C-H) deformation lig, car. [1,4]	all	1417-1421		increase	
1375	w, sh	δ(H-C-H) cel hem [1], C-H deformation cel	CD065	1369	sh	decrease	cel, hem decomp.
1327	m	C-H deformation cel, hem, ν(C-H) cel, ν(C_1-O) syringyl derivates [1]	all	1328-1330		decrease	cel decomp.

1263	vw, sh	C-H deformation (cel, hem), ν(C-H) cel, ν(C_1-O) syringyl derivates [1]	all	1265-1270		increase	
1223	m	ν(C=O) lig; ν(C=O) xyl [1]	all	1223-1227		decrease	lig, xyl decomp.
1157	w	δ(C-O-C) cel, hem [1], pol [4]	all	1150-1151	w, sh	decrease	cel, hem decomp., shift in δ(C-O-C) cel, hem [1]
absent			all	1122-1124	m, br	increase	ν(O-H) associations cel, hem [1], ring asymetric valence vibration pol [4]
1093	vw, sh	ν(O-H) associations cel, hem [1]	CD089, CD079, CD064	1090-1093		vw, sh	ring asymetric valence vibration pol [4], ν(O-H) associations cel, hem [1]
1034	vs	ν(C=O) cel, lig, hem [1]	all	1030-1043		decrease	cel, lig, hem decomp.
987	w, sh	ν(C=O) cel, hem [1]	CD065	987		vw, sh	cel, hem decomp.
926	w, sh	un-assigned, possibly ν(C=O), δ(R-C=O), or aromatic C-H out-of-plane bending of cel and/or hem and/or lig	all	914-924		increase & decrease	
899	s	(C-H) deformation cel [1], anomere C-groups C(1)–H deformation, ring valence vibration pol [4]	CD065	895	S	mostly decrease	





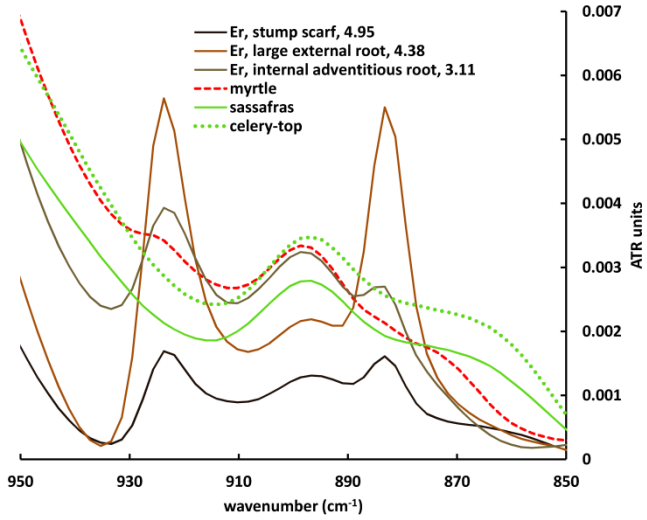


Figure A-II-3. Sections of the FT-IR spectra of different timber species for comparison with myrtle. The species were common ones in the mixed-forest of the Styx Valley, Tasmania. 'Er' is *Eucalyptus regnans* (swamp gum).

A-II.5 Discussion

The ages of the two samples of cunnite, ~200 and 300 years suggests that they correspond to the approximate age of the wood in their original myrtle trees. There is, as yet, no evidence that cunnite persists long beyond the complete decomposition of the original tree. However, for these two samples it has persisted while its surroundings have decomposed, to form a tree hollow, and become part of the atmosphere.

More data and/or a more in-depth analysis is necessary before the chemistry of cunnite is sufficiently determined so as to determine the apparently rare conditions for its synthesis in nature and to enable its synthetic manufacture. The most pertinent aspect related to the chemistry of cunnite is to determine either: (a) the difference in bonding in myrtle compared with other tree species, or (b) the difference in the brown rot fungi that consumes different species and in myrtle forms myrtle cheese. The spectra revealed that there was chemistry unique to myrtle, in comparison to the other timbers analysed. The reason for ability of myrtle to form conchoidal fracture upon decomposition with brown-rot fungi was not determined, but the greater range of bond types observed for myrtle than either the hardwood or softwoods may allow more chance of new bond formation upon decomposition. With a lignin content of ~16.7%, myrtle wood has a lignin content in the lowest 5% of tree species globally (Brauns and Brauns, 1960). As closest match of the cunnite spectra that of a primarily lignin-rich material then a considerable amount of other material must been decomposed. This would possibly allow a large degree of restructuring, with the possibility of new bonds forming between decomposition products and between those and remaining lignin.

The spectral analysis was not definitive in terms of being able to assign chemical bonds specific to either cunnite, myrtle wood or the other decomposition samples. There was qualitative evidence of decomposition of several wood components and evidence for persistence of, or new ones for, bonded water and hydrogen bonding.

But from studying the spectra it could not be determined quantitatively that more cellulose had decomposed than lignin.

The automated spectral matching showed that cunnite is primarily lignin and therefore the fungus that produces myrtle cheese is mostly a brown-rot fungus. Preston et al. (1998) show similar decomposition effects but with NMR rather than FT-IR and included the last decay stage, corresponding to lignomor, with ~60–95% of the remaining carbon belonging to lignin, depending on the tree species, with ~85% on average. With cunnite being primarily lignin it therefore has a slower decomposition rate than residue left after decomposition by white-rot fungi and represents a more-temporally stable carbon pool (Harmon et al., 2009).

The areas of in Tasmania that could potential contain cunnite are 0.077 and 1.00 Mha, in *E. regnans* mixed-forest and rainforest respectively, depending on their history of natural disturbance and logging. The Tasmanian rainforest species do not have a large export market in comparison to eucalypts. The un-merchantable myrtle trees (or so called 'useless trees' Florence and Shepherd (1975); Newbery et al. (1999)) are knocked over to prevent fire spreading from the intense 'regeneration' burns, and rainforest neighbouring a logging coupe can be used as a fire suppressant for otherwise uncontrolled silvicultural regeneration burns (Forestry Commission, 1978). Rainforest regeneration is suppressed by burning the humus mounds around large eucalyptus, where rainforest trees are concentrated, to prevent a second humus fire in the ensuing, young secondary forest (Cremer, 1962). In this context, it is important to note that myrtle cheese and cunnite form only in mature or senescent myrtle trees, which are absent from forest harvest cycles, though they constitute a minor carbon pool. The timber industry has been converting these primary forests for nearly a century and primarily exporting eucalypt pulpwood (The Mercury, 1941; ANM, 1979, Chapter 1). Scientific study on the non-industrial aspects of those primary forests has been sparse, and discoveries continue to arise from their remnants.

In NW Tasmania a portion of land assigned to the forestry industry, contained 107,000 ha of rainforest and 53,000 ha of mixed-forest in 1982, and it was assessed for myrtle pulping (FORTECH, 1982). There was 30 Tg (million tonnes) of pulpwood of which 86% was myrtle. A large and significant portion of the myrtle timber was found to have decayed to some degree but the potential woodchips were of acceptable quality. The recommendations were for a more precise inventory of the available pulpwood and 'a précis of likely public reaction'. With the current impetus to burn native forest 'logging residue' for energy, the assessed pulpwood stock could be designated as fuel. Recognition of the alternative properties of the myrtle timber could thereby give it a value in its current state and help prevent a substantial C emission from logging the primary forest. The work necessary in order to reveal more about the formation of cunnite and to develop and make use of its synthetic form requires further investigation of its chemistry.

Appendix III. Formulae in CAR4D

The formulae listed below are part of the forest C modelling software CAR4D as described in Dean et al. 2003; Dean et al. 2004; and Dean & Roxburgh, 2006. Many of the formulae used in CAR4D have previously been published but some have not. Some of the major formulae for tree growth and decomposition parts in CAR4D (previously published or unpublished) are listed here for completeness. CAR4D is designed for even-aged *Eucalyptus regnans*-dominated forest so unless otherwise stated the equations are for that species.

A-III.1 Diameter Distribution

Tree diameter as a function of age:

$$DBH_{ob} = a - [b \exp(c\ Age)]$$

R^2=0.82, N= 2184, MSE= 0.095, Eq AIII-1

where DBH_{ob}= diameter over bark at 1.3 m of one tree, Age= tree in years, a= 3.53(6), b=3.59(6), and c= -0.00323(8).

$$\ln(stocking\_rate) = a - [b \ln(stand\_Age)]$$

R^2=0.92, MSE= 0.39, N=82, Eq AIII-2

where ln is the natural logarithm, *stocking_rate*= number of trees per hectare, *stand_Age*= stand age in years, *a*= 10.6(2), and *b*= 1.27(4).

'age_env' (environmental age) is an adjustment of the real age in years to account for site index (local productivity), for which formulae are given in Dean et al. (2004). DBH and stocking rate are adjusted to suit age_env.

For each stand age a distribution of DBHs is assigned, from the lognormal distribution around the mean DBH from EqAIII-1.

$\sigma = 0.465651\exp(-0.032273 age\_env) + 0.307868$

$o\sigma sqrt2pi = 1/\left(\sigma\sqrt{2\pi}\right)$

$o2\sigma sqrd = 1/(2\sigma^2)$

The number of trees per hectare, which can be fractional with any particular DBH_i, is $numDBH_i$, where:
$numDBH_i = (o\sigma sqrt2pi/DBH_i)\exp(-diff^2 o2\sigma sqrd)$,
$diff = \ln(DBH_i) - \ln(DBH_{ob})$, and DBH_{ob} is given by EqAIII-1.

The DBH distribution for each age and environment is determined reiteratively. Limits for the range of DBHs within a year are estimated then occurrences that have too low a possibility of occurrence (≤1/30) are eliminated and the range correspondingly adjusted.
$max\_DBH = -0.1 + 0.14\exp(age\_env^{0.225})$, if max_DBH>12 then max_DBH= 12 .
The minimum DBH is adjusted for the environmental age. A list of DBHs is created between the upper and lower limits, according to the stand density for the environmental age. DBH values are separated by a single step size within the expected DBH range. The frequency of occurrence is calculated for each DBH and if less than 1/30 it is rejected and the distribution recalculated.

A-III.2 Biomass

For seedlings and very young saplings a realistic DBH cannot be assigned because they may be under 1.3 m height or 1.3 m up may be a stem tip above branches, which is not what is meant by DBH. Instead CAR4D uses ‘environment-independent’ tree volumes, based on a variety of literature sources but adjusted for environment (based on the parameters assigned by the user).

env_age (years)	stem_volume (m^3)	bark_volume (m^3)
1	0.000003550	0.000003485
2	0.00002915	0.00002188
3	0.0005571	0.0001767
4	0.0016231	0.0003767
5	0.003320	0.0005695
6	0.00671437	0.001102
≥7	equation below	equation below

Stem volume for each DBH in the DBH range for each env_age≥ 7 is:

$$stem\_volume = 1100\left\{1 - \left[{}^{1}/_{1 + (DBH_i/9.2)^2}\right]\right\}$$

with R^2= 0.95, N= 24 and RMSE= 34.2m^3, which incorporates cross-sectional area deficit. The ratio of stem density at present age to that for mature trees is:

$density\_ratio = 1 - \left(0.580915 \times \exp^{-0.2 \times age}\right)$, then dry stem biomass is:

$stem\_biomass = 512.4 density\_ratio\ stem\_volume$.

The derivation of the density of 512.4 kg m^{-3} is given in Dean et al. (2003). Bark volume in m^3 for each DBH in the DBH range for each env_age≥ 7 is:

$bark\_volume = -0.0000052 + 6.5\left\{1 - \left[{}^{1}/_{1 + ((DBH_i + 0.01)/3.7)^{1.8}}\right]\right\}$, and:

$bark\_biomass = 491.4 bark\_volume$.

An upper limit is placed on bark_biomass: not more than 0.05*stem_biomass.

The proportion of branch and leaf biomass is specified individually as a function of stem biomass for very young trees according to a range of literature sources including Feller (1980):

age_env	*live_branches*, multiple of *stem_biomass*	*live_leaf*, multiple of *live_branches*
1	0.3	1
2	0.25	0.4
3	0.20	0.2
4	0.17	0.0830663263
5	0.12	0.0830663263
6	0.11	0.0830663263
7	0.10	0.0830663263
8	0.09	0.0830663263
≥9	(0.05208 + 0.014309)/ (0.81631 + 0.11281)	0.0830663263

Stem volume and stem biomass are then adjusted for site index. Subsequently, stem volume, branches and leaves are adjusted for the decomposition accompanying senescence of *E. regnans* with age. This is dependent on senescence rates set by the user. Formula for the senescence adjustment are described in Dean et al. (2004). This gives a new value for "above ground biomass". After adjustment for the decomposition the root biomass is calculated:

$root\_biomass = above\_ground\_biomass \times \left(\left(3.0 \times \exp^{-0.85 \times age\_env}\right) + 0.15\right)$. With the value of 0.15 for root:shoot ratio being in between those of Feller (1980) and Mokany et al. (2006).) For understorey, root biomass was 10% of aboveground biomass.

The formula for understorey dry biomass was a blend from several literature sources (Gilbert, 1959; Ashton, 1976; Feller, 1980; Polglase and Attiwill, 1992; Brown and Nelson, 1993):

$$understorey\_biomass = bm\left\{1 - \left[{}^{1}/_{1 + (age\_env/350)^{k}}\right]\right\}$$

where mass is in tonne ha^{-1}, and bm = 1600. The understorey biomass is also scaled for site index using the same scaling factor as used for the stand's *E. regnans*, and a second scaling factor, ranging from 0.1-to-1 is available for the understorey at the user's discretion. A value commonly applied is 0.5, e.g. for a site of moderate productivity in the Styx Valley, Tasmania.

A-III.3 Biomass decomposition

The products of tree biomass decomposition were assigned to decomposition pools. Dead trees, due to self-thinning, were assigned to DBH-dependent decay pools with individual decomposition rates. Dead leaf fall for *E. regnans* was adapted from the constant rate reported by Polglase and Attiwill (1992) but with a rapid increase with age after 1 year:

$dead\_leaves = 2600\left[1 - \exp\left(0.4\sqrt{age\_env}\right)\right]$ in kg ha^{-1} for age_env> 1, and =0 for age_env= 1. The rates of decomposition of leaves and bark were adapted from Ashton (1975b):

$remaining\_dead\_leaves = dead\_leaves\_old\ exp(-0.5986431)$, where *dead_leaves_old* is the mass of leaf litter from the previous year, and values are in kg ha^{-1}, and

$remaining\_dead\_bark = dead\_bark\_old\ exp(-0.1321594165)$, where *dead_bark_old* is the mass of bark litter from the previous year, and values are in kg ha^{-1}. Understorey litter fall was adapted from Polglase and Attiwill (1992) and Ashton (1975b):

$understorey\_litter = 1500\left[1 - \exp\left(0.1\sqrt{age\_env}\right)\right] + 376\left[1 - \exp\left(0.1\sqrt{age\_env}\right)\right]$, where mass is in Kg ha^{-1}, and *age_env* ≥3. For *age_env*< 3, *understorey_litter*= 0. Decomposition of understorey litter was adapted from Turnbull and Madden (1986):

$remaining\_understorey\_litter = understorey\_litter\_old\ exp(-0.409)$, where values are in kg ha^{-1}.

Appendix IV. Controversies around logging: state of knowledge and rebuttals

This appendix is adapted from two published papers: Dean (2011) and Dean (2012).

A-IV.1 Abstract

Moroni et al. (2010) (doi:10.1155/2010/690462) reported extant, spatially representative carbon stocks for Tasmania's State forest. Their disputation of earlier work, contextual setting, redefinition of carbon carrying capacity (CCC), methods, adoption of ecological concepts and consequent conclusions on carbon flux were investigated. Their reported data were very useful, however the absence of sufficient context and of fundamental equations were atypical of scientific publications; oldgrowth should have been differentiated from other mature forests, and wet-sclerophyll from mixed-forest; redefinition of CCC was unwarranted; and several of their arguments and conclusions appeared unwarranted. From the graphs and tables in their paper I estimated that the carbon deficit in State forest biomass (the amount below CCC) due to commercial forestry was conservatively 29(±4) Tg (or 106(±13) Mtonnes CO_2-eq; with couped-production forests 29(±6)% below CCC), which is a greenhouse gas mitigation opportunity— indicating the usefulness of the existing definition of CCC. Also using their data, earlier work on long-term fluxes accompanying conversion of wet-eucalypt forests to harvesting cycles was found to correspond to 0.56(±0.01) Mha (more than a third of the State forest), 76(±2)% of which is in the commercial production area— in contrast to their claim that earlier work referred to a small and atypical proportion.

A-IV.2 Introduction

There are some long-term controversies in the literature regarding carbon accounting of forest logging and rangeland management. Forest logging is considered in this chapter and rangelands are considered in the following chapter. The point of

contention regarding logging is whether or not the carbon balance is negative or positive, i.e. does it contribute to climate change or does it constitute climate change mitigation. Some of the most topical carbon pools are wood-products and soil organic carbon (SOC). However the scene is muddied by differing approaches to the dimensions of the pools, the categorisation of C within them, and dissonance between the derivation and application of data.

It would be remiss to not acknowledge the divisive ambience that pervades Australian society, especially in Tasmania, with regards to existing large industries and carbon emissions, climate change, industrial land use and conservation (in its broadest sense). It is part of a global struggle in which environmental science is subverted by the large industries that stand to lose from conservation(Bradshaw and Ehrlich, 2015). Climate change denial, industries' and conservationists' politics and public relations, corporate defences against environmentalists, government alignments, forestry blockades, animal rights activism, intentional environmental damage and human fatalities are all involved in such conflicts (e.g. Clarkson, 1964; Kirkpatrick, 1998; Kirkpatrick, 2000; Hanson and White, 2003; Ajani, 2007; McCright and Dunlap, 2010; Krien, 2012; Bayldon et al., 2013; Sneddon et al., 2014; Winter and Poole, 2014; Woodburn, 2014; Bradshaw and Ehrlich, 2015; Fuller and Razaghi, 2015; Medimorec and Pennycook, 2015; Vidot and Conifer, 2015). The sentiments driving the dichotomy are long-standing:

> "FOREST PRESERVATION.
> To the Editor of the West Australian.
> Sir,—I do not hear of late any of our public men alluding to a future attempt being made to conserve our forests, although all men I speak to on the subject—beyond a few who are materially interested in the present system—agree that it is a matter of the utmost national importance and something should be done. We may look at our sister colony, South Australia, and see what strides have been made there in this direction, and a very little calculation will put us in possession of the fact that their apparently expensive system will one day be a national benefit of no mean commercial and climatic value. But we are in a forty fold better position to enter upon a system productive of immediate returns than is South Australia, or even any other division of Australasia, and, simply because of this fact, our legislators are apparently lethargic, and say, "Oh ! it will last our time, our forests are boundless," and so on. A very little research will show us the mistake we are labouring under. If we refer to America we may find that where originally there was fertility and a fair proportion of forest, with the loss of the latter the former has been turned almost into barrenness; and America once said, as we do, "Our forests are everlasting." We have indigenous to our colony the best two woods known in the southern hemisphere for general purposes, the "Tewart" (Eucalyptus Gomphosephylla), and "Jarrah" (Eucalyptus Marginata), and, taking this into consideration alone, I, for one, hold that something

should be done at once to prevent the reckless waste daily going on in our forests, and to assist the reproduction of these woods. I further hold that a system may be established that would be of no material hindrance (commercially) to our lumberers* and still be of vast importance to the State.

Trusting you will pardon my intrusion on your space,

I am, etc.,

SYLY1A.

*Lumberers-An American term for saw millers, wood merchants, etc."

The West Australian, Monday 29th June 1885, page 3

The work in this thesis has been influenced by groups and individuals with anti-environmental intentions, pro-conservation intentions and by those leaning neither way. For example, much of the literature studied for this thesis, providers of equipment, and people facilitating its progress, were neutral. However, the Wilderness Society of Australia provided citizen science data; Forestry Tasmania permitted me access to locked forests and their library, and it published rebuttals of papers emerging from the present work; and the forest industry lobby group, Institute of Foresters of Australia, contacted me directly, indicating disapproval. Although the categorisation is not black-and-white, the fundamental causes for the difference between anti- and pro-environment intentions, have been established to some degree (e.g. Dunlap, 1975; Milfont, 2007; Karpiak and Baril, 2008; Feygina et al., 2010; Devine-Wright et al., 2015) and therefore a solution to the dichotomy could theoretically be deduced. Here, I simply exhibit some of my responses to controversies, neutral or otherwise, on carbon dynamics of LUC by forest logging. Although anti-environmental involvement stifles progress in applied environmental science and indirectly depletes the environmental global commons, it prompts rebuttals that promote study of processes that would otherwise come under the heading of pure science, a type of science presently unpopular, though nevertheless with merit. Consequently it is worth reporting the subsequent rebuttal work here. Another reason for exhibiting the rebuttals is in case people believed industries' responses represent reality. The topics covered here are the overall carbon balance of primary forest logging, and the effect on SOC from logging. The aim of this Appendix is: to provide the rebuttal of some of the industry rebuttals as clearly as possible, and to present the larger context so as to allow more-productive science.

Tasmania is the State of Australia with the climate and soils most-suitable to high-biomass forests (per hectare). It is the leading State exporter of wood-products, mostly pulpwood for fine paper production (Attiwill and Adams, 2008; ABARE and BRS, 2010), and it has had multi-decadal discord over forest usage, requiring repeated Federal intervention and economic support. One recent controversy is accounting for the carbon footprint of the native forest industry. Forestry Tasmania (FT), the direct descendant of the Forestry Commission of the 1970s and earlier the Forestry Department, has responsibility for the management of wood production in a large proportion of Tasmania's forests.

Moroni et al. (2010), employees of FT, report extant, spatially representative carbon stocks for the native forests and non-forest areas in Tasmania's public 'State forest' of 1.5 Mha. They dispute earlier work on carbon (C) dynamics, propose a new definition of Carbon Carrying Capacity (CCC), and advocate specific management options. Correspondingly, the stated aims of Moroni et al. (2010) were (1.) to 'improve understanding of forest C stocks at the landscape level' and (2.) to 'estimate the Carbon Carrying Capacity of Tasmanian State forest and explore the methods required to achieve and maintain Carbon Carrying Capacity'.

The presentation of acreages and carbon densities of forest types in Moroni et al. (2010) is self-contained and fulfils their first aim to a substantial degree. However, they add arguments without supporting data change definitions, and ignore historical context. They dispute earlier work on forest carbon dynamics and provide a subjective application of ecological concepts and related management options. In order to best progress related science, it is necessary to scrutinise, clarify and correct several issues apparent in Moroni et al. (2010). These are summarised in Table A-IV-1 and discussed below.

Table A-IV-1. Summary of issues in Moroni et al. (2010) addressed in this chapter

Issue	Clarification and adjustment provided
Insufficient context to display C flux science	Description of the state-of-the-science, including types of C stocks and fluxes needed to address the knowledge gaps in commercial forestry, applicable to [global] climate change science.
Earlier work contested and disputed	Logic and explanation to show context, validity and significance of earlier work.
Redefinition of CCC	Validity of the existing definition of CCC, its relevance to appropriately addressing the knowledge gap, and why their redefinition is inappropriate.
Methods and Results section	Explanation of data standards and methods necessary for scientific reproducibility in this field, to promote more useful reporting and more usability of data.
Discussion and Conclusions sections	Ecological context allowing interpretation the C dynamics. Explanation of how their results do not support some of their discussion and conclusions.
Additional Realisations	CCC and C deficit calculated [from their data] to meet the stated aim and for climate relevance. Quantified validation of contested earlier work.

A-IV.3 Context and literature survey

In the introduction and discussion in Moroni et al. (2010) the authors claim that earlier work on carbon accounting in southeast Australia inappropriately:

(a) focused on high biomass forests
(b) did not include landscape-level effects such as fire or different stand ages
(c) incorrectly interpreted and calculated CCC

However the reporting of the state-of-the-science by Moroni et al. (2010) misses key aspects such as the context of global climate change; the forest industry's historical context regarding C flux; requirements for climate change research; and the earlier

spatiotemporal work such as age-related calculations, fire-impact modelling and landscape-level, temporal forecasts.

A-IV.3.1 Context of C flux research

Carbon accounting of extractive forestry activities is relevant to climate change science (as part of LUC) and to ecology (e.g. through biogeochemical cycles). With respect to climate change it helps quantify the degree attributable to anthropogenic greenhouse gas (GHG) emissions (Meir et al., 2006), with contributions from land use necessary for quantification of climate change positive feedback (Gloor et al., 2010); more generally it can determine industry's carbon footprint. Moroni et al. (2010) began by stating that 'Storing carbon (C) in forests' is the reason for C flux analysis of 'forested landscapes'. But that is only one aspect— namely that pertaining to emission offsets— measuring young specimens and assuming a blank slate. A major part of the science (e.g. for calculation of current and future climate change) is calculating benefits from conservation of extant forest C stocks such as those in primary forests (in biomass, necromass and soil), a field which entails analysis of industry-induced C fluxes (e.g. Harmon et al., 1990; Harmon et al., 1996); assessment of long-term, pre-industry stocks; and thus determining the amount of emissions [to be 'offset'], i.e. the industry carbon footprint. As 25–50% of anthropogenic emissions remain in the atmosphere for 500–10,000 years (Houghton et al., 1994; Archer et al., 2009; Eby et al., 2009), improved accounting for both historic and future emissions aids climate change modelling. Apart from direct reduction in forest biomass, other emissions from resource-extractive forestry, such as decomposition of wood-products and changes in soil carbon are all strong contributors to anthropogenic GHG emissions and their quantification is currently poor (Houghton, 2008). From the 1960s in southeast Australia, wood production from native forests increased and pulpwood export dominated (Chapter 3, section 3.2). Accounting for that period and for future activity, relates directly to climate change, and entails flux determination and historical and future CCC determination.

This crucial setting explains the function of the earlier work disputed in Moroni et al (2010).

Moroni et al. (2010) suggest that the early work on southeast Australia, by looking at the more-carbon-dense forests and older-age classes, was misplaced. However, the Tasmanian mature, primary wet-eucalypt forests have been and continue to be, a prime source of high quality sawlogs and export pulpwood (Helms, 1945; Felmingham et al., 2004; Elliot et al., 2008). Much of that resource extraction from the 1960s onward involved clearfelling, intense burning (with high emission from logging slash) and then either reseeding with native seed (Hickey, 1994) or plantation development. Presently, the annual area felled of dry-eucalypt, (with lower C density per unit area than wet-eucalypt), outnumbers that of wet-eucalypt but extraction from the drier forests began later and was at a slower rate (McCormick and Cunningham, 1989). Also the more-mature rainforest (e.g. >200 years) on more fertile soils and the older mixed-forests have already been sourced for the higher-priced, deep-red myrtle sawlogs (TWFF, 2004). Addressing such major influences on biomass, necromass, and soil by the forest industry inherently requires study of the high-biomass and older forests.

The mathematical reason for analysing older forests and larger trees, is to facilitate calculation of an ample expanse of any growth curve (Dean, 2003), these being necessary for: (1.) forecasting carbon trajectories under different management and wildfire scenarios (e.g. Dean and Roxburgh, 2006; Roxburgh et al., 2006b), (2.) for determining how the extant soil carbon stocks were acquired (a result of the long half-lives of some soil pools), (3.) for calibrating biogeochemical models, and (4.) for modelling various ecological processes. Most work to-date on forest mass has been attuned to timber production, with less attention to elemental or compound fluxes (e.g. C, N and H_2O) and little to environmental consequences of the timber trade. Mass and volume research has focussed on forecasting timber outputs, especially where investment was highest, such as in silvicultural stands. For example, in the comprehensive review of biomass allometrics for southeast Australia (Keith et al., 2000), of 78 formulae, 53 were for plantations and 25 were for native

forests. Of those for native forests the minimum and maximum DBH were 0.16(0.10) and 0.64(0.55) metres respectively (SD in brackets) (N=24) and the maximum stand age was 67(66) years (N=21), these being far below dimensions of mature and oldgrowth specimens (e.g. Maiden, 1908; Table 1-1). Thus, allometrics prior to 2000 had included comparatively few mature stands although mature primary forests are the source of most native forest wood-products in southeast Australia, and are consequently expected to be the source of high C fluxes. New research must attend to knowledge gaps in the existing science. Indeed, from a Commonwealth government project to implement Australia's National Carbon Accounting System, it was found that more data were needed for high-biomass forests (Richards and Brack, 2004). It was in that vein that the work of Keith et al. (2000); Dean (2003); Dean et al. (2003); Keith et al. (2009) and (Roxburgh et al., 2006b) began, as part of the Cooperative Research Centre for Greenhouse Accounting. It was a founding step in advancing the science, and its contestation by Moroni et al. (2010) neglected the scientific and public-relevance contexts.

A-IV.3.2 Spatio-temporal aspects disputed in the literature critique in Moroni et al. (2010)

Regarding temporal representativeness, in the series of papers looking at Tasmanian harvesting by the current author (from Dean et al. (2003) to the present thesis), the long-term averages of pre-logging C stocks (which included all growth stages, from seedling to senescence), and the long-term harvesting-cycle average stocks (including wood-products) were used to calculate logging effects (namely relative change in carbon stocks per unit area for conversion of native forest to harvesting cycles). Thus they were temporally representative for that forest type. Time-based average is a method recommended by the IPCC for accounting for change in soil carbon: 'best estimated over several rotations or disturbance cycles' (IPCC, 2003), and it is equitable to also apply it to the biomass and necromass pools over the same accounting period in native forests.

Additionally, in Chapter 4 it was shown that such difference in long-term stocks were unaltered if the first logging of the native forest was prior to maturity (e.g. at 60 years) or when it had become mature— a consequence of using long-term averages— and thus the results were to some degree also representative of that forest type across the forest estate. Therefore the statement of Moroni et al. (2010): 'estimates of Carbon Carrying Capacity in the above studies are demonstrated as the difference between current forest carbon stocks and those anticipated for an area when supporting solely mature, carbon-saturated forests', is incorrect, and misleading.

As a first attempt to represent State-wide fluxes associated with extractive forestry in Tasmanian wet-eucalypt forests, in Chapter 2, E1- and E3+-type forest (the latter an uneven-aged, mixed-species forest, low-biomass forest, once subject to light selective logging but recovered sufficiently to oldgrowth status) were modelled— finding approximately the same relative emissions for both forest types upon conversion to harvesting cycles. With regard to State-wide representation of forest growth stages and detailed forest-typing— Forestry Tasmania [the affiliation of Moroni et al. (2010)] are the custodians of that data and it is not publicly available in electronic form— some printed maps are available but they require scanning and digitising before use in a GIS. For C flux research in the absence of the regional, digital data, approximations must be made.

Also, although Moroni et al. (2010) cited Dean et al. (2003) and Dean and Wardell-Johnson (2010) they did not acknowledge the corresponding landscape-level and spatio-temporal work of Dean et al. (2004) and Dean and Roxburgh (2006) (although the latter is referenced on a research website administered by Forestry Tasmania). The two spatio-temporal papers included modelling of catchment-level C forecasts, including all growth stages, showing model sensitivities to gaps in the ecological knowledge and they presented modelling of different fire management scenarios at the catchment-scale. They did not attempt to measure CCC, but nevertheless included both natural and unnatural disturbance by including forest regrowth from the 1800 and 1939 fires in Victoria, the latter being anthropogenic in origin (Stretton,

1939). Thus it was spatio-temporally representative. Therefore the statement of Moroni et al. (2010) 'discussions of forest C in Australia to date either lack a landscape view or have derived a landscape view from sites representing a small and atypical proportion of the forest landscape' is incorrect; and from the issue outlined above regarding major, industrial C fluxes, their statement neglects the fundamental reason for 'discussions of forest C'.

A-IV.4 Actual issues in earlier work

There were however errors in some of the papers which Moroni et al. (2010) disputed (although not identified in their critique) and it is relevant to summarise them: (a) in Dean et al. (2003) the loss of soil carbon with each disturbance event was an overestimate when considering the full soil profile and harvesting area— corrected in Chapter 2 and in Dean and Wardell-Johnson (2010); (b) similarly, Mackey et al. (2008) misplaced soil carbon emissions that might accompany long-term commercial forestry activities, as short-term emissions— noted by Roxburgh (2009); and (c) in Table I in Dean and Wardell-Johnson (2010) the potential carbon in biomass from FullCAM (Richards and Brack, 2004) should be halved as they were reported in terms of C and not the dry-biomass of the original dataset, and the title changed to indicate that they were for aboveground biomass only (corrected in Chapter 2)— values are then closer to those of Keith et al. (2010) who noted that the FullCAM layer underestimates potential of wet-eucalypt C by approximately half, and the FullCAM values are even further from those found in Chapter 3 and Chapter 6— further strengthening the point about the FullCAM data in Chapter 2. However, none of those errors in earlier work impacted on the calculated emissions from biomass and necromass (including decomposing wood-products) accompanying commercial forestry.

A-IV.5 The redefinition of CCC

Moroni et al. (2010) suggested that the basic concept of CCC was inapplicable to the forest estate; they suggested it was incorrectly calculated by Roxburgh et al. (2006b), Mackey et al. (2008) and Keith et al. (2010), and that CCC needs redefining as a theoretical estate-wide maximum, independent of natural disturbance, that can never be achieved. Moroni et al. (2010) incorrectly cite Nabuurs et al. (2007) when they say: 'Such calculations estimate the theoretical biological maximum forest carbon stocks achievable or Theoretical C Saturation [11].' The statement to which they refer in Nabuurs et al. (2007) was: 'The theoretical maximum carbon storage (saturation) in a forested landscape is attained when all stands are in oldgrowth state, but this rarely occurs as natural or human disturbances maintain stands of various ages within the forest.' Moroni et al. (2010) ignore that the forests measured in Roxburgh et al. (2006b) and in Keith et al. (2009) showed fire impacts. For example, various fire types were mentioned in Roxburgh et al. (2006b), and the Keith et al. (2009) study highlighted uneven-aged stands of *E. regnans*, which is a [fire] obligate seeder. The methods used in relation to wildfire effects and a range of stand ages in those earlier CCC calculations are amply explained in a CSIRO review of sequestration and GHG mitigation opportunities (Roxburgh, 2009). Although wildfire was not a focus in Chapter 4 it was part of the modelling, such as when calculating long-term average C stock, and it was covered in earlier papers (Dean et al., 2004; Dean and Roxburgh, 2006). The fact that carbon stocks would be lower immediately after a landscape fire (and possibly lessen the C balance of logging), does not affect the long-term emissions for LUC of primary forest, as by definition, it includes fires at all scales. A different outcome to the calculations in Chapter 4 may occur if the forests are frequently burnt prior to logging but then the forests and C forecasts would be of a vastly different type, more like dry-sclerophyll forests.

The CCC work of Keith et al. (2009) was an initial estimate. Therefore future work requires more spatially and temporarily representative measurements of CCC— an area of ongoing research (personal communication, Brendan Mackey, 2011). It is possible that the work of Keith et al. (2009) required data on both higher- and lower-

biomass stages of southeast Australian forests but that is insufficient reason to redefine CCC. The situation was best summarised by Roxburgh (2009) in his analysis of the Keith et al. series of papers on CCC: '...it requires knowledge of the actual spatiotemporal variation in natural disturbance history of the forest estate, the spatiotemporal variation in other attributes relating to site quality, and the site histories from where the data were collected.'

The landscape saturation to which Moroni et al. (2010) refer was not used for landscape-level calculations nor for industry carbon footprint calculations in the Dean et al. series, Roxburgh et al. (2006b) and the Keith et al. series, as suggested by Moroni et al. (2010). It was only used for unit-area (point or site) calculations at specific points in time, which is appropriate. CCC, although not named as such, has been applied to the USA forest estate and incorporates natural disturbance and reveals the opportunity for sequestration due to the deficit created by timber harvesting (e.g. Brown et al., 1997; Albani et al., 2006; Potter et al., 2008).

Moroni et al. (2010), after rejecting and redefining CCC, then calculated and used theoretical State-wide C saturation, and stated that it was an impossible scenario. From their data they did not present a calculation of CCC or an industry C footprint, nor did they discuss relevance of their data or results to either climate change mitigation or adaptation. The relevance of their findings to climate science was not stated.

It must also be noted that it is not possible in southeast Australia to measure the highest-biomass forests possible at the regional-level, because of LUC after European colonisation (section 1.1.2 in Chapter 1). For example, the amount of Tasmanian rainforest burnt anthropogenically was much higher in the 20th century than in the previous two centuries and has included much attrition of primary forest (Kirkpatrick, 1994). Also, current firewood extraction from Australian native eucalypt forests is ~7 Tg year^{-1} of biomass (Driscoll et al., 2000). Consequently, wherever CCC is derived from an observation of the upper limit of present-day stocks, it could well be an underestimate. Therefore it is possible that a landscape-

level CCC calculated from extant stocks, but without representative-inclusion of very young regenerating stands, could offset the absence of the missing high-biomass stands, and thereby inadvertently result in a reasonably accurate estimate of CCC.

In their discussion section, Moroni et al. (2010) unnecessarily segregated two concepts: CCC and wildfire. After their redefinition of CCC as 'theoretical carbon saturation' in the introduction, they suggest in the discussion that earlier calculation of CCC was invalid because wildfire reduces landscape-level C stocks from the level of C saturation and they imply that human-induced fires are natural. (However, wildfire may not always reduce C stocks in biomass in the short-term if it is not of stand-replacing severity, as evidenced by the high biomass of uneven-aged stands of fire-sensitive species.) The standard definition of CCC, which they quoted in the introduction, includes: '...and natural disturbance regimes, but excluding anthropogenic disturbance', and therefore already provides sufficient distinction of events. In Australia's Kyoto Protocol obligations, forest emissions from arson are classified as "natural" (e.g. DCC, 2008), however, for scientific purposes (such as in climate change modelling) the distinction must be made between 'natural' and 'anthropogenic'. Around 50% of Australian forest fires are ignited by arson but it can be as high as 75% (Forestry Tasmania, 2001; Bryant, 2008). There are also fuel-load reduction burns, specifically in forests, to prevent arson (State Forests NSW, personal communication, 2009). Moroni et al. (2010) suggest that because of wildfire (in which they include anthropogenic fire) then their redefined CCC is an impossible concept, with the implication that a measure of CCC is impossible. However, wildfire is intrinsic to both the standard definition of CCC and to the time-based average method of determining carbon budgets for forest activities.

Additionally in their discussion Moroni et al. (2010) state that their redefined CCC cannot be reached because some locations have low fertility or water availability. However the earlier work that they contested inherently included such environmentally variability in site potential, through avenues such as remote-sensing, growth adjusted for site index, and a range of study sites (e.g. Dean et al., 2004;

Keith et al., 2010). This further negates the need for their redefinition of CCC and disputation of the earlier work.

A-IV.6 The methods and results sections

Three major detractions from an appropriate-level of reporting appear in the Methods and Results sections in Moroni et al. (2010):

(a) the definition of mature forests did not suit calculations for CCC nor for spatiotemporal-specific carbon calculations,

(b) neither sufficient methods nor data were provided for the standard reproducibility of a scientific paper, and

(c) error margins were not presented for some forest types and for the penultimate calculations.

Moroni et al. (2010) defined the upper limit of biomass, as stands at ≥110 years of age. That would under-represent any specific-site's potential biomass, and regional CCC, because mixing of biomass values for all the ages ≥110 years would produce a low value compared with potential. The age of 110 years may well represent 'maturity' for timber harvesting purposes but it does not represent ecological or C maturity— the latter being crucial to such accounting studies.

For example, from CAR4D (Dean and Roxburgh, 2006) the estimated total C in biomass of an *E. regnans* mixed-forest, type E1.M+ (one of the types targeted for pulpwood and sawlogs), at 110, 200, 300 and 400 years (including losses from senescence) is 402, 594, 768 and 864 Mg ha^{-1} respectively. The time-based averages of C in biomass for that forest type from 110–200 years and 200–400 years are 506 and 754 Mg ha^{-1} respectively, i.e. 50% higher for oldgrowth than for mature to early-oldgrowth. Such mixed-forests can still have high biomass in their eucalypt component when they are beyond 500 years of age (Mount, 1964; Wood et al., 2010). Similarly oldgrowth stands of ~250 years of age in the Carpathian Mountains had 50% more biomass than mature stands of 100–150 years (Keeton et al., 2010).

However, the average-aged wet-eucalypt C in biomass in Moroni et al. was 222 Mg ha^{-1}— only 4% lower than their 'mature' value of 232 Mg ha^{-1}, and indistinguishable within error margins.

The spatiotemporal representativeness of oldgrowth in the stands of ≥110 years of age in Moroni et al. (2010) was not stated. Together with the extant landscape-level stocks being below CCC values due to extensive logging activity of high-biomass forests and individual trees in Tasmania (Chapter 4) then a regular grid array of sampling points would not measure representative oldgrowth stands. Representative oldgrowth stands may not have been included in the analysis of Moroni et al. (2010), which would result in their potential biomass value for forests of ≥110 years being unrepresentative for several of the forest types they measured, and in-turn, unrepresentative of landscape-level potential, i.e. of CCC. Nevertheless, their values for some specific sites appear to have included oldgrowth forests, and their values for some forest types appear to be within expected values.

Reproducibility is the core of the scientific method, 'it enables scientists to evaluate the validity of each other's hypotheses and provides the basis for establishing known truths' (Gil et al., 2007). For example, the International Union of Crystallography has a long-standing requirement that observed and calculated structure factors are deposited with publication of structures— thus enabling alternative structure refinements. There are two main methods for biomass calculation in temperate forests: stem volume estimation and destructive sampling. Moroni et al. (2010) apparently used the stem volume method via 'a variety of protocols' (plus biomass expansion factors) and a third method of 'expert opinion'. Allometrics relating readily measureable physical attributes of the tree to stem volume can be achieved from taper formulas, remote sensing, or mill measurements. Moroni et al. (2010) did not present the allometrics they used, to show how stem volume was achieved from DBH and height data and therefore their methods are not reproducible.

Expert opinion may be satisfactory for a qualitative paper but not for a quantitative one, especially one where conclusions are based on quantitative data such as Moroni

et al. (2010), and where the underlying reason for the research is to provide numeric input to another field, namely climate science. Steps used in the 'expert opinion' and 'various protocols' in such a report must be presented, at least as supplementary material. Additionally, Forestry Tasmania (to which the authors of Moroni et al. (2010) are affiliated) has privileged access to the data and to the forests and therefore has the responsibility of disclosure to the scientific community. The reason for retention of that information may be that it is commercial-in-confidence. However, much of that information was probably acquired prior to commercialisation of the government department. Also, as mentioned in Moroni et al. (2010), Forestry Tasmania is the legislated administrator of the State forests' resources and therefore has no business competition. Sufficient formulae were provided in the Dean et al. series of papers, Roxburgh et al. (2006b) and in the Keith et al. series.

The penultimate values presented for total extant C stock in biomass were not assigned error margins. This may have been because the rainforest components (classes 85 and 86) did not have error margins assigned. Error margins could have been calculated in part from their Table 2 and separately for the 'expert opinion' estimates. Without error margins then their conclusions based on comparisons have reduced weight.

A-IV.7 Ecological concepts and consequences in the discussion and conclusions sections

In their Results section Moroni et al. (2010) calculated a comparison of wet-eucalypt forest with rainforest, namely their classes 4 and 5 compared with classes 85 and 86. But the reason and validity for that comparison were not stated and the error margins associated with their rainforest biomass were not stated (although they were for other forest classes) — obscuring the validity of the comparison. In their discussion section that change of forest types [to rainforest] was extended to cover all wet-eucalypt forests being converted to rainforest. However, in their methods section, regarding non-eucalypt forest, they stated: '...standing-tree volumes are derived from expert

opinion because little tree measurement data is available', and 'little to no data was available for noneucalypt dominated forests or non-merchantable biomass components of eucalypts. These data gaps require further study...'. However, a substantial portion of their discussion and conclusion was based on the transition of wet-eucalypt forest to rainforest, which requires going through the intermediate stage of mixed-forest, the mature-to-senescent stages of which has a significant portion of rainforest species biomass, for which they had 'little to no data'. With that lack of data for myrtle and sassafras (a major component of M+ rainforest biomass), their quantitative and qualitative conclusions of lower biomass for rainforest than for wet-eucalypt forest— e.g.: 'rainforest C density is expected to be much lower'— appear unwarranted.

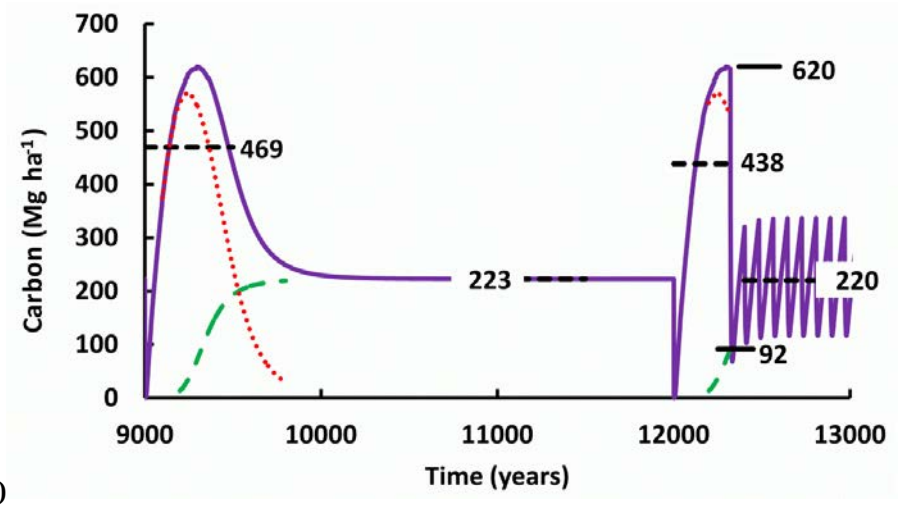
In Chapter 2 it was shown that, at the landscape-level, modelled data for both the biomass and SOC are higher in the rainforests of Tasmania then in its wet-eucalypt forests. That estimate for biomass was based on the FullCAM layer (Richards and Brack, 2004) which used productivity, soils, climate and remote-sensing data. Thus it was more geographical than temporal, and it did not necessarily represent differences in biomass from catchment-scale forest succession. Nevertheless it remains as a possible guide to relative biomass. The remote-sensing data from which the FullCAM layer was derived, was two-dimensional, and therefore did not differentiate biomass between different stem heights (e.g. with eucalypts being taller than rainforest trees). Alternatively, rainforest species, being more shade tolerant, can have higher stand densities and thus possibly acquire more biomass below a shorter canopy than can eucalypts. Until more data are revealed, the conclusion of a net drop in biomass through the seral succession of wet-sclerophyll to mixed-forest to rainforest cannot be made. Also, the soil carbon pool contributes to total carbon for each forest type and that would also have to be included in a C comparison of forest types.

There is insufficient evidence to unequivocally support the proposition that C stocks will be lower if eucalypts could die out and be replaced by rainforest because the net C flux for the transition, for example as calculated in May et al. (2012a), are usually

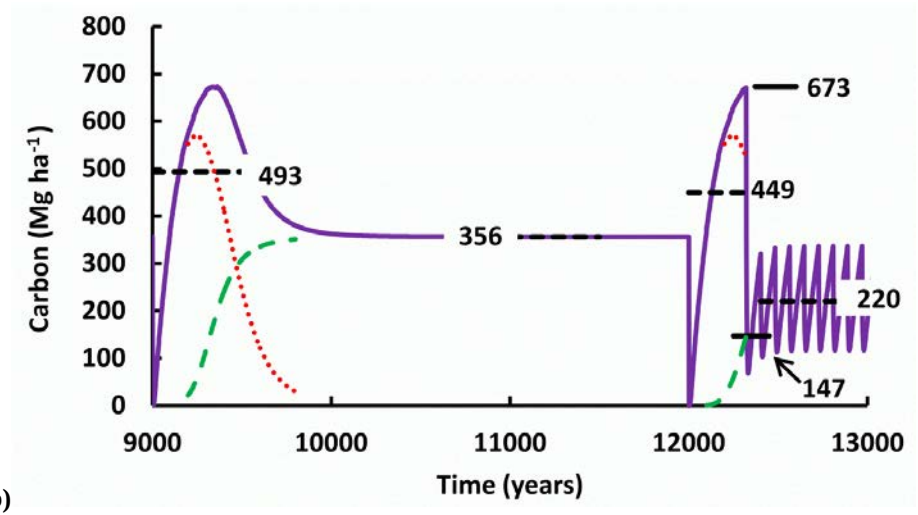
based on data from stands at the late stages of eucalypt senescence, and therefore do not reflect the possibility of carbon accumulation in rainforest after the eucalypts die out. Furthermore, climate change is likely to induce additional fire and there were ample stands of wet-eucalypt forest prior to the first industrial logging in Tasmania.

To illustrate possible biomass dynamics of rainforest succession and to compare these against logging, two simulations were run using CAR4D (Figure A-IV-1), with two levels of understorey biomass: one very low and one higher but still only at around 20% of total stand biomass at the time of logging (Chapter 3). Rainforest understorey can contribute 50% of total biomass prior to onset of substantial eucalypt senescence in mixed-forest (Green, 2002). Due to the low precision of understorey allometrics, no definitive quantitative result is shown in Figure A-IV-1, but even when including wood-products, it is not necessarily the case that harvesting cycles store more carbon than long-term rainforest.

The long-term biomass and CWD carbon levels in wet-eucalypt forests oscillate with fire, giving a long-term mean below that of mature primary forest (Figure A-IV-1). Long-term rainforest does not have that oscillation. Consequently, the long-term C stock of biomass in rainforest can exceed long-term eucalypt logging-cycle carbon (including wood-products) if it is a third or more of the biomass of mature wet-eucalypt primary forest (Figure A-IV-1.a). The C in biomass of rainforest understorey only needs to be about 15% of the C in biomass of primary forest at the time of logging, if the two forest types are to be equivalent in terms of C. However, long-term rainforest carbon needs to be more than half of peak mixed-forest carbon, or more than 20% (as understorey) at the time of logging of primary forest, if it is to substantially exceed long-term mixed-forest carbon (Figure A-IV-1.b).



(a)



(b)

Figure A-IV-1. Simulated forest succession scenarios, as might be typical of Site-1 (Chapter 4), showing biomass plus wood-products for primary-forest and harvesting cycles (using CAR4D). (Solid curve=total biomass plus wood products; dashed green curve=understorey biomass; dotted red curve=*E. regnans* biomass. Dashed horizontal lines=time-based averages; solid horizontal lines=carbon stock at time of logging.) The rainforest understorey was allowed to fill occupancy in the senescing mixed-forest stand and then to succeed to rainforest for 2000 years, after which there is an intense wildfire with seeding from nearby *E. regnans*. The new mixed-forest grows to age 320 years, at which time it is logged (parameters as in Chapter 4). Rainforest biomass was set to a maximum of ~1/3 (a) and ~1/2 (b) that of the peak of the mixed-forest— these corresponded to 15% and 22% (respectively) of the stand-level biomass at time of primary-forest logging. In (a) the long-term rainforest biomass is about half that of the time-averaged mixed-forest biomass and equal to that of the harvest cycles (including wood-products). In (b) long-term rainforest biomass is close to that of the time-averaged mixed-forest biomass and greater than that of the harvesting cycles (including wood-products).

Mention of biomass estimates of mixed-forest as a separate category, are absent from the paper of Moroni et al. (2010) although they may be included within some of the 'mature' wet-eucalypt categories, E1, E2 and E3+. They are however delineated in State forest mapping by Forestry Tasmania (Stone, 1998). Development of mature rainforest components in mixed-forests (e.g. myrtle, a common component of M+ type mixed-forest) requires a fire-free period of about 200 years (Hickey, 1994; Tabor et al., 2007). The biomass of oldgrowth mixed-forests, being a superposition of two forest types, is likely to be higher than that of each component separately, unless pure rainforests can exceed the eucalypt biomass in mixed-forest, or where tall eucalypts replace rainforest in some wet-eucalypts forests. A variety of C trajectories for related scenarios are portrayed in the Dean et al. series of papers. More precisely, change in biomass with forest succession for a wide range of forest types in New Zealand is detailed in Hall and Holinger (2000): most often the early-seral stage (the pioneer stage) has higher biomass, although not always, which is in contrast with the claim of Moroni et al. (2010) that reduced biomass is 'atypical ecology'. In the absence of fire in the eastern USA mixed-hardwood forests (*Acer* spp., *Quercus* spp and *Liriodendron tulipifera* L.) can be replaced by the more shade-tolerant Eastern Hemlock (*Tsuga canadensis* (L.) Carr.), with increasing C density. There will be a range of such succession possibilities in Tasmania. The different rainforest types and eucalypt habitat may be linked to interactions between fire frequency and soil chemistry (Read, 2001). The spatio-representativeness of the different eucalypt types and associated rainforest types in the mixed-forests would need to be assessed before concluding on one carbon trajectory State-wide, as in Moroni et al. (2010).

Commercial forestry, through its conversion of primary wet-eucalypt forests, together with some rainforests, to eucalypt forests with an average harvesting cycle well below 200 years, has increased the proportional coverage of rainforest-free eucalypt stands— thereby decreasing the likelihood of occurrence of both mixed-forest and rainforest. That situation is likely to continue with to the planned forestry activities in Tasmania over the next two to three decades. Thus there is little danger

of a significant loss of biomass by conversion of wet-eucalypt to rainforest over the next century or two, even if the rainforest and mixed-forest had lower biomass than the rainforest-free wet-eucalypt forests.

Moroni et al. (2010) implied that without anthropogenic intervention a significant portion of wet-eucalypt forest would become rainforest, but there is no evidence for that scenario. If it was indeed probable then there would be little mixed-forest or wet-sclerophyll forest present today. With climate change the forests of Tasmania are forecast to change to drier forest types (Chapter 2) which implies rainforest succeeded by mixed-forest then by wet-sclerophyll etc. The concept of 'drought-free areas' in Tasmania raised in Moroni et al. (2010) is unrealised, as average annual precipitation, whatever level, does not preclude drought. Fire severity and frequency is expected to increase with climate change (Chapter 2) and direct anthropogenic fire in Tasmania is not expected to decline, but it increases logarithmically with population (Venevsky et al., 2002). Thus there is no evidence to suggest a reasonable likelihood for the concluding scenario portrayed in Moroni et al. (2010) of conversion to rainforest with accompanying loss in State-wide C stocks. Instead their data could have been used to assess effects of commercial forestry, to-date and into the future, or the effects of climate change.

A-IV.8 Additional realisations

A rough estimate for CCC for State-forest in Tasmania can be derived from the data presented in Fig. 3 of Moroni et al. (2010). As an initial and conservative estimate one can assume that the only C deficit exists in the wet-eucalypt forests (as they have been the most targeted to date (e.g. Chapter 4)) and only in the areas of even-aged silvicultural regeneration (i.e. for E1or2 and E+3orX, ≤1959-to-2000s; within classes 50-to-74). Other deficits across the production area of State forest are: (1.) non-clearfell logging (of eucalypt and rainforest); (2.) the 83800 ha of State plantation (MBAC, 2007); (3.) anthropogenic fire and firewood collection; (4.) the deficit in E3-, E4 and E5 type (dry-eucalypt) production forests; and (5) the logging roads.

Data extracted from Fig. 3 in Moroni et al. (2010) had a procedural error margin of approximately plus or minus one line width, i.e. ±1.2 Mg ha^{-1} for C densities, and ±200 ha for the areas: i.e. a maximum of ~0.5%. Empirical error margins in C density were reported in Table 2 in Moroni et al. (2010). The potential C densities for regenerating, wet-eucalypt forests were calculated from the area-weighted C densities of forests types E1, E2 and E3+, at age ≥110 years, with and without regrowth (i.e. amongst forest classes 1-to-17): 348(±40), 259(±20) and 188(±17) Mg ha^{-1} respectively. Inclusion of the forests containing regrowth (which were of lower C density than without regrowth) and the lower-aged mature forests (≥110 years), rather than using data for oldgrowth alone, made the estimated potential more conservative and represented temporal variability, i.e. it corresponded more to a time-based average. The potential [time-based average] stock for the wet-eucalypt, silvicultural regeneration forests is then simply their areas multiplied by the potential C densities. The C deficit is the difference between those potential stocks and the extant stocks: 29(±4) Tg (Table A-IV-1). The CCC, at the Sate-wide level (across all native forests on State-owned land, including those in reserve) is the extant stock for native forests plus the C deficit: 193(±23) Tg. The areas of regeneration would not all achieve maximum C stock simultaneously; nevertheless, as the potentials were not derived from oldgrowth data alone but from data for 'age ≥110 years, with and without regrowth' forests, then the deficit and CCC are spatiotemporally representative.

Table A-IV-1. Carbon deficit and CCC for State-forests based on data in Moroni et al. (2010), accounting for wet-eucalypt only. Values were rounded to three significant figures for presentation but not during calculations. a: '110wawr'= age≥110 years, with and without regrowth.

#	Carbon pool	Source	C density or stock	Area (Mha)
A	Potential, E1or2, 110wawr[a]	Fig. 3	303(±28.7) Mg ha^{-1}	
B	Potential, E+3orX, 110wawr	Fig. 3	188(±17.3) Mg ha^{-1}	

C	Extant, E1or2 even-aged regrowth	Fig. 3	7.39(±0.813) Tg	0.0950
D	Extant, E+3orX even-aged regrowth	Fig. 3	2.22(±0.245) Tg	0.0520
E	Extant, native State-forests	Fig. 3	164(±2) Tg	1.38
F	Potential, E1or2 even-aged regrowth, 110wawr	Area(C)×A	28.8(±2.87) Tg	0.0950
G	Potential, E+3orX even-aged regrowth, 110wawr	Area(D)×B	9.78(±0.948) Tg	0.0520
H	Deficit, E1or2 plus E+3orX	F+G-(C+D)	29.0(±4) Tg	0.147
	CCC, native State-forests	E+H	193(±23) Tg	1.38

An independent estimate for the C deficit, with which to compare the value of 29(±4.0) Tg, can be calculated from MBAC (2007), a report commissioned by Forestry Tasmania to provide a carbon budget for 2007–2057. If native forest logging had ceased in 2007 and the commercial estate could sequester C at the same rate as the non-commercial estate, then from Fig. 8 in MBAC (2007) the State forest biomass would sequester an additional 30(±2) Tg of C by 2057 (after discounting the ~6 Tg of C for changes in soil, debris and wood-products pools associated with logging). This would suggest that the C deficit value of 29(±4.0) Tg calculated above from data presented in Moroni et al. (2010) is of the expected order of magnitude and is likely to be conservative (as the sequestration modelled (MBAC, 2007) would most likely continue beyond 2057). Note that these figures do not incorporate impacts on future growth due to climate change.

From Fig. 5 in Moroni et al. (2010) the area of production forest was 0.871(±0.009) Mha, and that of 'couped production' forest was 0.570(±0.006) Mha. The total C in production forest and couped-production forest from Fig. 5 was 93(±2) and 71(±2) Tg respectively. With the 29(±4) Tg deficit calculated above for even-aged regeneration of E1, E2 and E3+ type forests, the production and couped-production forests in Tasmania are 24(±5)% and 29(±6)% respectively below their CCC— that

is where management has most opportunity for sequestration ventures. (Note: (a) these percentages ignore the other deficits, as mentioned above; and (b) they do not assume State-wide C saturation, but they do assume that all the commercial forestry has been in the couped-production forests.)

The area referred to in the calculations in Chapter 2, as the area subject to 50% emissions of carbon stocks in the long-term, upon conversion to harvesting cycles, is that which contains forest types E1, E2 and E3+ (i.e. wet-eucalypt forests). From Fig. 3 in Moroni et al. (2010) the area of those forest types was 0.56(±0.01) Mha; from their Fig. 5, this is equivalent to 38(±1)% of the whole State forest area, and combining that acreage with their partitioning in their Table 3 indicates 76(±2)% of that forest type is in the production area and 9(±1)% is in formal reserve. Those representations of 38(±1)% and 76(±2)% indicate that the statement in Moroni et al. (2010) suggesting that areas 'capable of growing such very tall eucalypt forests ... comprise only a small proportion of total forest area' is thus unwarranted. Additionally, major emissions with commercial forestry are relative to the differences in long-term, time-based-average C densities (Chapter 2, Chapter 4) and therefore both area and C density are of relevance to climate science, not area alone.

In summary the numbers presented in the Moroni et al. (2010) on C densities and acreages for forest types are very useful to science and carbon accounting. The portrayal of the state-of-the-science of forest C dynamics by Moroni et al. (2010) missed major aspects, such as the context, and was potentially misleading. The absences of sufficient context and of fundamental equations were atypical of scientific publications. Much of the discourse in Moroni et al. (2010) was a product of their unnecessary redefinition of CCC and their use of C saturation at a landscape-scale rather than its usual usage over a specific duration for a specific forest stand; similarly for their postulation of State-wide conversion of wet-eucalypt to rainforest. The reason for their train of arguments was not stated in their paper. From their graphs and tables: the estimated C deficit in Tasmanian State forests (the amount below its CCC) due to commercial forestry is currently 29(±4) Tg (or 106(±13) Tg CO_2-eq), with the couped-production forests 29(±6)% below CCC. That amount can

be sequestered for climate change mitigation, and the deficit shows the usefulness of the existing definition of CCC. Earlier work on long-term fluxes accompanying conversion of wet-eucalypt forests to harvesting cycles was found to correspond to 0.56(±0.01) Mha, (i.e. over a third of the forest estate), 76(±2)% of which is in the commercial production area— in contrast to their claim that earlier work referred to 'a small and atypical proportion of the forest landscape'.

A-IV.9 Response to further comments

Moroni et al. (2012) replied to the above content in this appendix (i.e. to Dean, 2011). They made numerous claims that if left unrefuted might mislead all but the most expert reader. There are forty claims which misrepresent my reply to their earlier paper. Seven of the most important of these are addressed below.

On page 2 Moroni et al. (2012) unexceptionally state that '...Dean [2] acknowledges wildfire may be "intrinsic to both the standard definition of CCC and to the time-based average method of determining carbon budgets for forest activities,…".' However they follow this with the claim: 'this is then ignored in the incorrect generalisation of CCC values across the landscape'. I had calculated the estate-wide (i.e. landscape-level) carbon deficit due to the logging of a particular forest-type and age-class (that which has been most sought after— mature wet-eucalypt primary forest) and compared it with an independent value of the estate-wide deficit from MBAC (2007). The calculated deficit was then used to reveal a conservative value for the estate-wide CCC (conservative as other forest types were also logged but not included in the calculation). The method was correct because it used the two actual states of the forest (with and without logging), as observed. If any wildfire had occurred since logging, the regenerating stands would have been re-categorised and removed from the particular data set presented in Moroni et al. (2010) and hence the deficit I calculated using these figures was due to logging alone.

I used a range of mature stand ages in my calculation of the pre-logging carbon, and hence also in the carbon deficit and in the calculation of CCC (Table A-IV-1). Note though, that using a range of stand ages as I did for the pre-logging state of wet-eucalypt forests yielded a conservative value of the carbon deficit for those forest-types, because the logging has predominantly been of the more-mature subset of primary forest (e.g. The Mercury, 1938; The Mercury, 1941; Helms, 1945; Felmingham et al., 2004; Elliot et al., 2008) which can have 50% higher biomass than early-mature stands (Appenidx-IV.6).

On page 2 of Moroni et al. (2012) they state: '...Dean [2] appears to agree when he criticises Moroni et al. [1] for "their unnecessary redefinition of CCC and their use of C saturation at a landscape-scale rather than its usual usage over a specific duration for a specific forest stand. ...Dean himself applies CCC at the landscape scale when he writes "the estimated C deficit in Tasmanian State forests (the amount below its CCC) due to commercial forestry is currently 29 (±4) Tg."' Moroni et al. (2012) thus inferred that 'the best example of the flaw in the logic' of my paper was the application of the concept of C saturation at the landscape-scale. I did not apply 'C saturation' at the landscape-scale. I used CCC at the landscape-scale.

Moroni et al. (2012) refer to a 'confusion between site-level CCC and attainable landscape-level C stocks' but after reviewing the literature I could only find such a 'confusion' in work by Moroni et al. (2010), Moroni et al. (2012) and Moroni Moroni (2012). For example in Moroni et al. (2012):

'The dynamic nature of these ecosystems and their C stocks in time and space prevents all stands across a landscape reaching stand-level CCC at the same moment, prevents any stand from remaining at its CCC indefinitely, and makes a nonsense of the goal of managing for stand-level CCC at the landscape scale [19]. It would be preferable to use the term theoretical carbon saturation *sensu* Nabuurs et al. [20] at the landscape level, restricting CCC to the stand or site level.'

No one would ever suggest that forest stands stay at CCC at a fine time scale, neither individually nor in unison. Nevertheless, by the most commonly applied definition of CCC, both stands and landscapes can, over appropriate time periods, maintain CCC— that is how any value of CCC is calculated— it is a spatio-temporal average.

In their second sentence cited here they suggest using C saturation at the landscape level— which would surely be a purely abstract concept only, and of little use, even theoretically.

On page 3 of Moroni et al. (2012) they make two claims about my use of forest class data: 'Dean [2] overestimates CCC primarily by basing his estimate on Forest Classes 1–17.'. For my CCC estimate (Table A-IV-1) I used a combination of: (a) data for those forest classes 1-17, (b) for their regrowth stage (classes 50-74), (c) an estimate of mature primary-forest C typical of those stands prior to State-wide high-grading, and (d) an estimate from an independent publication (MBAC, 2007). I restricted the classes used to even-aged regrowth and wet-eucalypt forests because they have been the most targeted to date (Chapter 4) and the required supplementary data for them is more readily available. If I had included an estimate of CCC and the carbon deficit for other logged forest types such as dry-sclerophyll and rainforest then the calculated carbon deficit would have been larger, rather than smaller; and the CCC higher.

On page 3 of Moroni et al. (2012) they suggest that carbon accounting would be better improved in agricultural and urban lands than in forests. However, the carbon balance of native forest industries remains very relevant to climate change science (Chapter 1).

While I did say that 'the Tasmanian forest industry has received repeated economic support...' (Moroni et al., 2012 p. 4), I did not make the 'suggestion' that the industry is financially unviable. I stated that the industry had received repeated economic support from the federal government in the context of showing how it related to other States and to the broader society, over a long time span. I was referring to the financing of industry restructuring towards the harvesting of plantations but also to such things as the diesel fuel rebate; federal government support for investment through compulsory, Australia-wide superannuation schemes; and the $2 million to develop alternatives to clearfelling oldgrowth forest (DAFF, 2010b).

On page 4 of Moroni et al. (2012) they imputed a 30% drop in my calculation of changes in soil carbon, with each logging event. The default change in soil organic carbon used with each clearfell, burn and sow event was 2.5% (Dean and Wardell-Johnson 2009). They also made a point of stating that this publication was 'unpublished'. But it is common practice in science to present findings at conferences in order to garner feedback from peers. – No complaints were received and those calculations have since been through double-blind peer review and published in a reputed journal (Dean et al., 2012d). Although the initial findings were presented at a conference they were publicly available on the internet, like several reports by Forestry Tasmania and Moroni, referred to by Moroni et al. (2010) and Moroni et al. (2012).

On page 4 of Moroni et al. (2012) they imply that I believe the carbon in wood-products to be irrelevant. I specifically referred to some of the Dean et al. series of publications where the wood-product pool is an intrinsic part of the context, some of which were cited in Moroni et al. (2010) and Moroni et al. (2012). The carbon deficit calculated above (Table A-IV-1) was carbon that could be sequestered in the forest. Wood-products cannot sequester carbon, as they cannot photosynthesise, rather they can at best temporarily behave as if inert but most often gradually decompose (or are burnt), thereby emitting C. I did not mention wood-products at length as the paper was about how to properly calculate the carbon carrying capacity of the State forests, rather than about the carbon budget as a whole.

Appendix V. Supplementary information for Chapter 10

Table A-V-1. Sources of NSW rangeland data.

Data type	Source	Format and Resolution
deforestation (from LANDSAT)	NSW Office of Environment and Heritage	raster, 25 m pixels
land-use	NSW Office of Environment and Heritage	polygons (resolution unspecified)
ecosystem-type (Mitchell Landscapes)	NSW Office of Environment and Heritage	polygons (resolution unspecified)
extant woody biomass (from ALOS radar), as of 2009	A research product developed by the Joint Remote Sensing Research Program (University of Queensland, Qld DERM, NSW OEH), the University of Aberystwyth (Wales, UK) and the Queensland Herbarium within the framework of the JAXA Kyoto and Carbon Initiative (© JAXA/METI)	raster, 50 m pixels
rainfall, temperature	Australian Bureau of Meteorology	raster, 0.025º (~25 km) pixels
tenure	Geoscience Australia	polygons, ~1 metre coordinate resolution
farm productivity	NSW Department of Primary Industries	polygons (resolution unspecified)
Interim Biogeographic Regionalisation for Australia, v6.1 (IBRA)	Department of Environment, Australian Government	Polygons (resolution unspecified)
soil organic carbon to 0.3 m	(Barson et al., 2002)	raster, 0.025º (~25 km) pixels

Table A-V-2. Categories and areas of land-use in NSW selected for analysis of commercial rangeland grazing.

Attribute 'LU_NSWdeta' in the GIS file LanduseV1.shp	Area (ha)
Grazing of native vegetation. Grazing of domestic stock on essentially unmodified native vegetation	1855454.57
Grazing of native vegetation. Grazing of domestic stock on essentially unmodified native vegetation - with a woody vegetation cover of mallee woodland	7.35
Grazing of native vegetation. Grazing of domestic stock on essentially unmodified native vegetation - with a woody vegetation cover of open forest	16293.83
Grazing of native vegetation. Grazing of domestic stock on essentially unmodified native vegetation - with a woody vegetation cover of woodland	256693.97
Grazing of native vegetation. Grazing of domestic stock on essentially unmodified native vegetation - with more than 30% of ground area having native shrub regeneration	23995.42
Grazing of native vegetation. Grazing of domestic stock on essentially unmodified native vegetation - with more than 30% of ground area having regeneration of native tree species	29788.04
Grazing of riparian land - with a woody vegetation cover of open forest	95.04
Grazing of riparian land - with a woody vegetation cover of woodland	174.43
Grazing within an ephemeral wetland - with a woody vegetation cover of closed forest	337.12
Grazing within an ephemeral wetland - with a woody vegetation cover of open forest	1711.35
Grazing within an ephemeral wetland - with a woody vegetation cover of woodland	7283.59
Grazing within an ephemeral wetland - with more than 30% of ground area having native shrub regeneration	486.49
Grazing within an ephemeral wetland - with more than 30% of ground area having regeneration of native tree species	134.12

Grazing within an ephemeral wetland (does not include cropping within an ephemeral lake - see classes 182 & 189)	54547.95
Grazing within bed of an ephemeral lake or watercourse; lake or watercourse are not regulated or above regulation level	464445.12
Grazing within bed of an ephemeral lake or watercourse; lake or watercourse are not regulated or above regulation level - with a woody vegetation cover of woodland	5590.90
Grazing within bed of an ephemeral lake or watercourse; lake or watercourse are not regulated or above regulation level - with more than 30% of ground area having regeneration of native tree species	887.97
Grazing within bed of an ephemeral lake or watercourse; lake or watercourse are not regulated or above regulation level with a dense shrub or tree cover	13610.93
Grazing within bed of an ephemeral lake; lake is regulated	10306.66
Grazing within controlled flood management systems	4.26
Rangeland grazing	27291359.55
Rangeland grazing - with a woody vegetation cover of open forest	2104.33
Rangeland grazing - with a woody vegetation cover of woodland	99355.51
Rangeland grazing - with more than 30% of ground area having native shrub regeneration	32148.34
Rangeland grazing - with more than 30% of ground area having regeneration of native tree species	64440.21
Rangeland grazing within western river outflows (eg Paroo & Warrego Rivers)	176999.04
Rangeland grazing; evidence of previous cultivation	2092.17
Recently cleared land (cleared of forest vegetation as yet not covered by crop or pasture)	15889.80
Secondary grassland in forested areas	13.04
Volunteer, naturalised or improved pastures - with a woody vegetation cover of open forest	1136.15
Volunteer, naturalised or improved pastures - with a woody vegetation cover of open woodland	196.87

Volunteer, naturalised or improved pastures - with a woody vegetation cover of woodland	12715.88
Volunteer, naturalised, native or improved pastures - with more than 30% of ground area having native shrub regeneration	20082.88
Volunteer, naturalised, native or improved pastures - with more than 30% of ground area having regeneration of native tree species	110619.49
Woodland (unmodified native vegetation)	20.04
Total	30571022.43 (~31 Mha)

Table A-V-3. Distribution of C change for different rangeland land-cover, for areas with biomass below the remnant-ecosystem biomass.

Land-use with ecosystem extant C stocks below remnant values	Change in C in aboveground biomass (Tg)	Percentage of Total
Grazing of essentially unmodified native vegetation	-13.633	23.3
Major riparian and wetland	-1.149	1.88
Rangeland grazing, various amounts of woody cover	-44.174	72.4
Recently cleared	-0.189	0.31
Secondary grassland in forested areas	-2.305E-04	3.80×10^{-4}
Volunteer, natural, native or improved pasture with some woody cover	-1.870	3.06
Woodland (unmodified native vegetation)	-5.499E-05	9.01×10^{-5}
Total	-61.01	100

Table A-V-4. Distribution of C change for different rangeland land-cover, for areas with biomass below the extant, local ecosystem-average biomass.

Land-use with ecosystem extant C stocks below extant average values	Change in C in aboveground biomass (Tg)	Percentage of Total
Grazing of essentially unmodified native vegetation	-5.4243	14.4
Major riparian and wetland	-1.0221	2.71
Rangeland grazing, various amounts of woody cover	-29.3202	77.6
Recently cleared	-0.004241	0.0112
Secondary grassland in forested areas	-2.577E-04	6.8×10^{-4}
Volunteer, natural, native or improved pasture with some woody cover	-1.9989	5.29
Woodland (unmodified native vegetation)	-1.1924E-04	3.16×10^{-4}
Total	-37.77	100

Table A-V-5. Recent deforestation (1989–2007) of commercial rangeland. Note that the 'land-use', although rangeland grazing, was sometimes named differently in the original data layer.

Land-use (from LANDSAT Identification)	Area (ha)	Emitted C in aboveground biomass (Mg)	
		NCAS	ALOS
Pasture	72,342	1,698,616	1,384,848
Infrastructure	10,021	193,620	146,299
Thinning	1040	37,843	34,851
Possibly agriculture	521	14,134	12,370
Previously missed clearing	400	11,526	9,467
Native forest logging	123	5,717	5,397
Plantation estab.	40	1,734	1,629
Settlement	0.39	9	9
Total	84,495	1,963,198	1,594,870

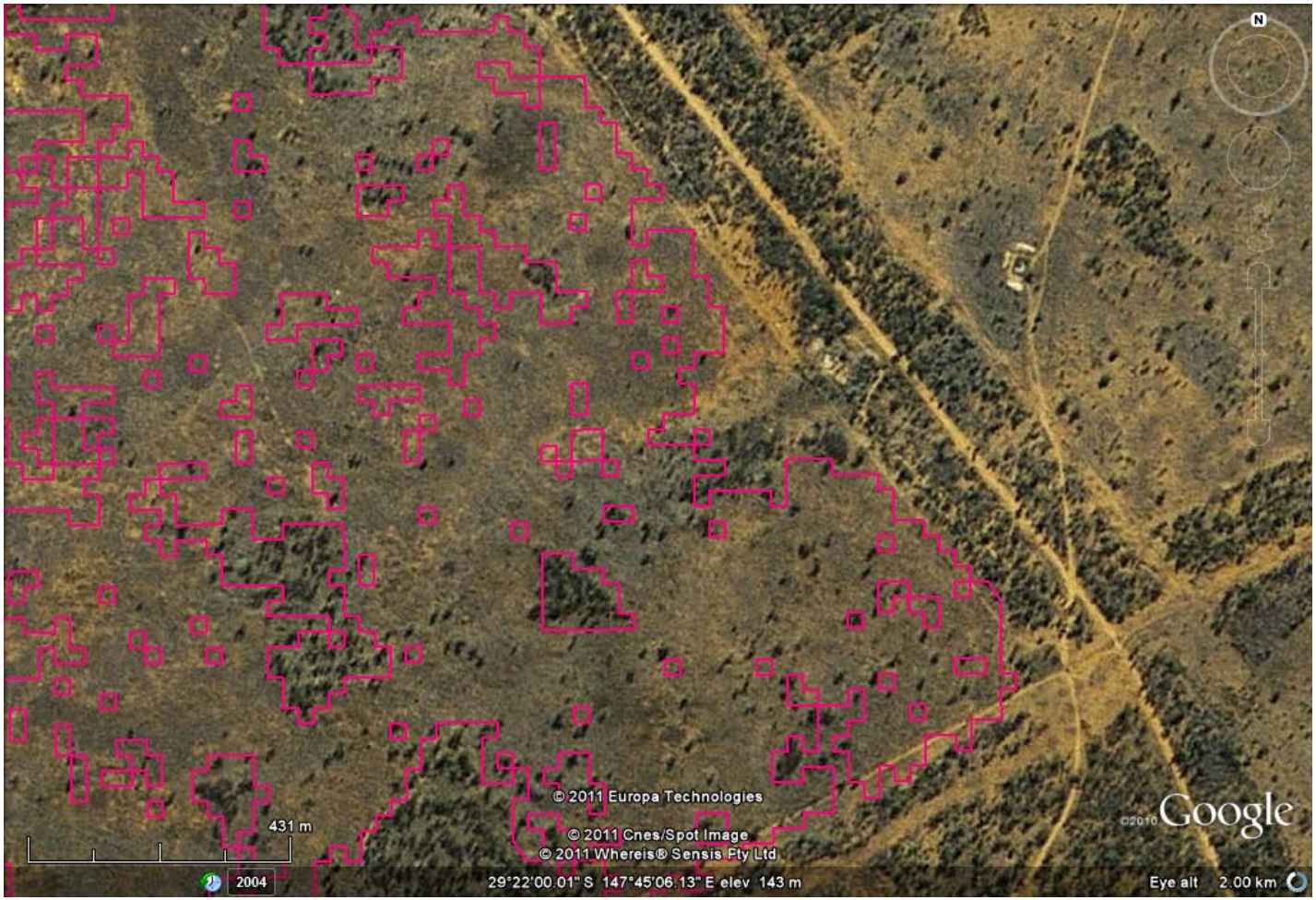
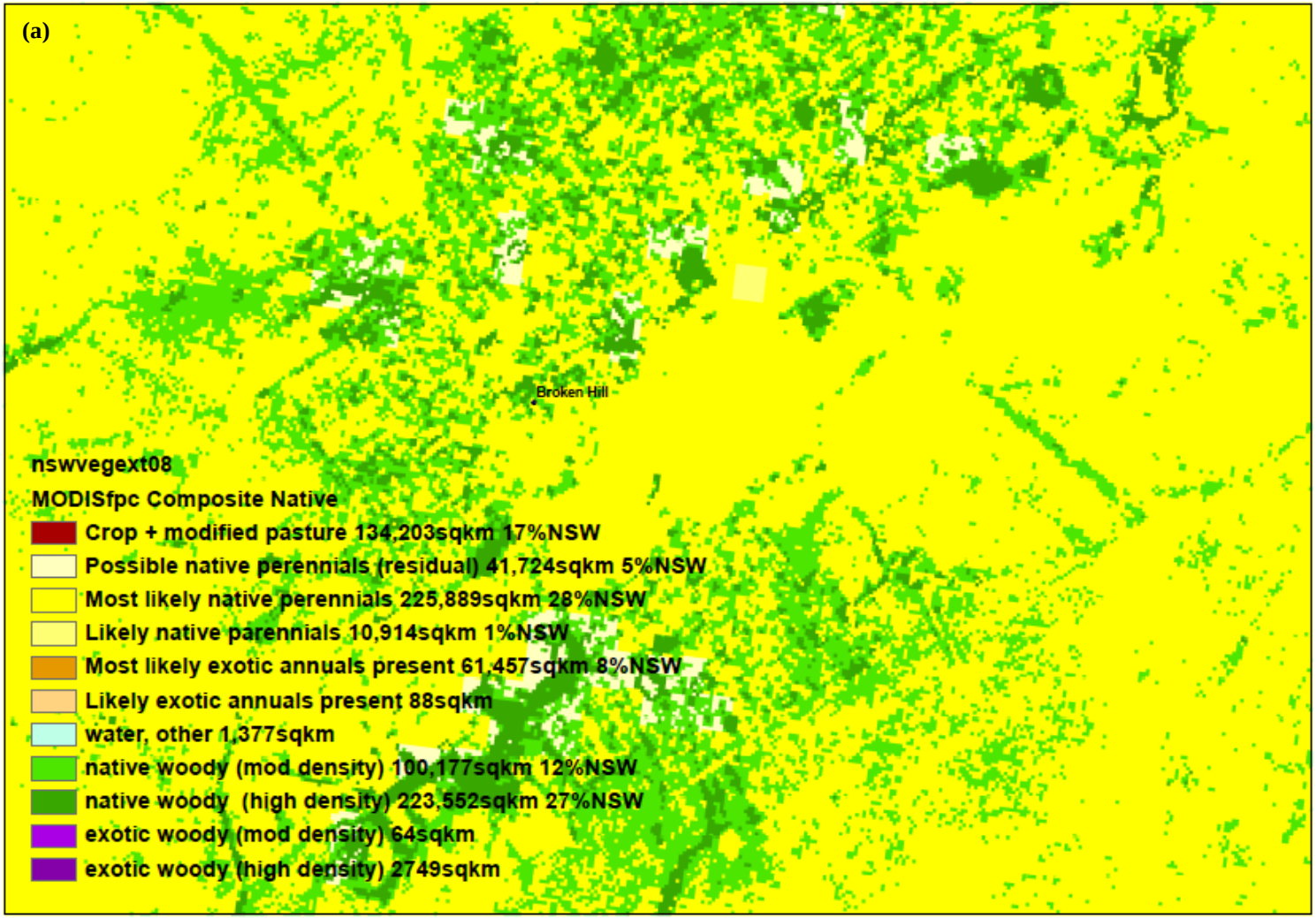


Figure A-V-1. Example of delineated deforestation (pink polygon outlines) derived from LANDSAT data overlaid on a GoogleEarth© image, near Lightning Ridge. Delineation had imperfections, with some deforestation undetected and some single trees being delineated as deforested.


(a)
Broken Hill
nswvegext08
MODISfpc Composite Native
Crop + modified pasture 134,203sqkm 17%NSW
Possible native perennials (residual) 41,724sqkm 5%NSW
Most likely native perennials 225,889sqkm 28%NSW
Likely native parennials 10,914sqkm 1%NSW
Most likely exotic annuals present 61,457sqkm 8%NSW
Likely exotic annuals present 88sqkm
water, other 1,377sqkm
native woody (mod density) 100,177sqkm 12%NSW
native woody (high density) 223,552sqkm 27%NSW
exotic woody (mod density) 64sqkm
exotic woody (high density) 2749sqkm

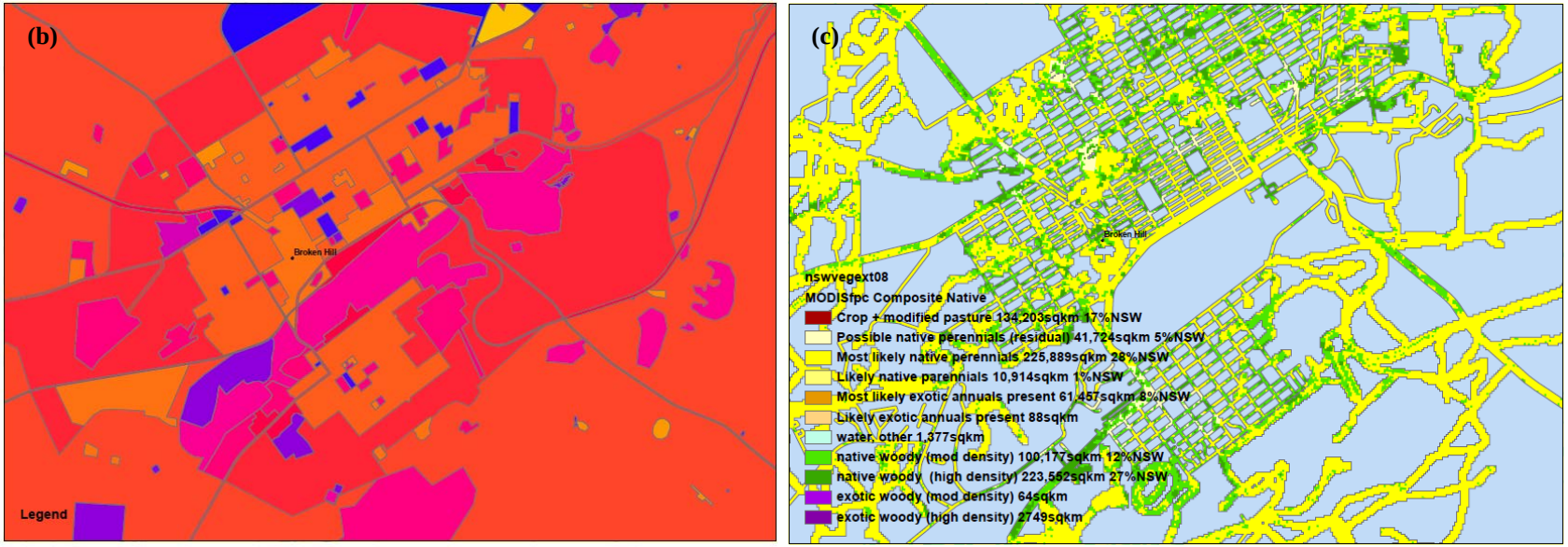
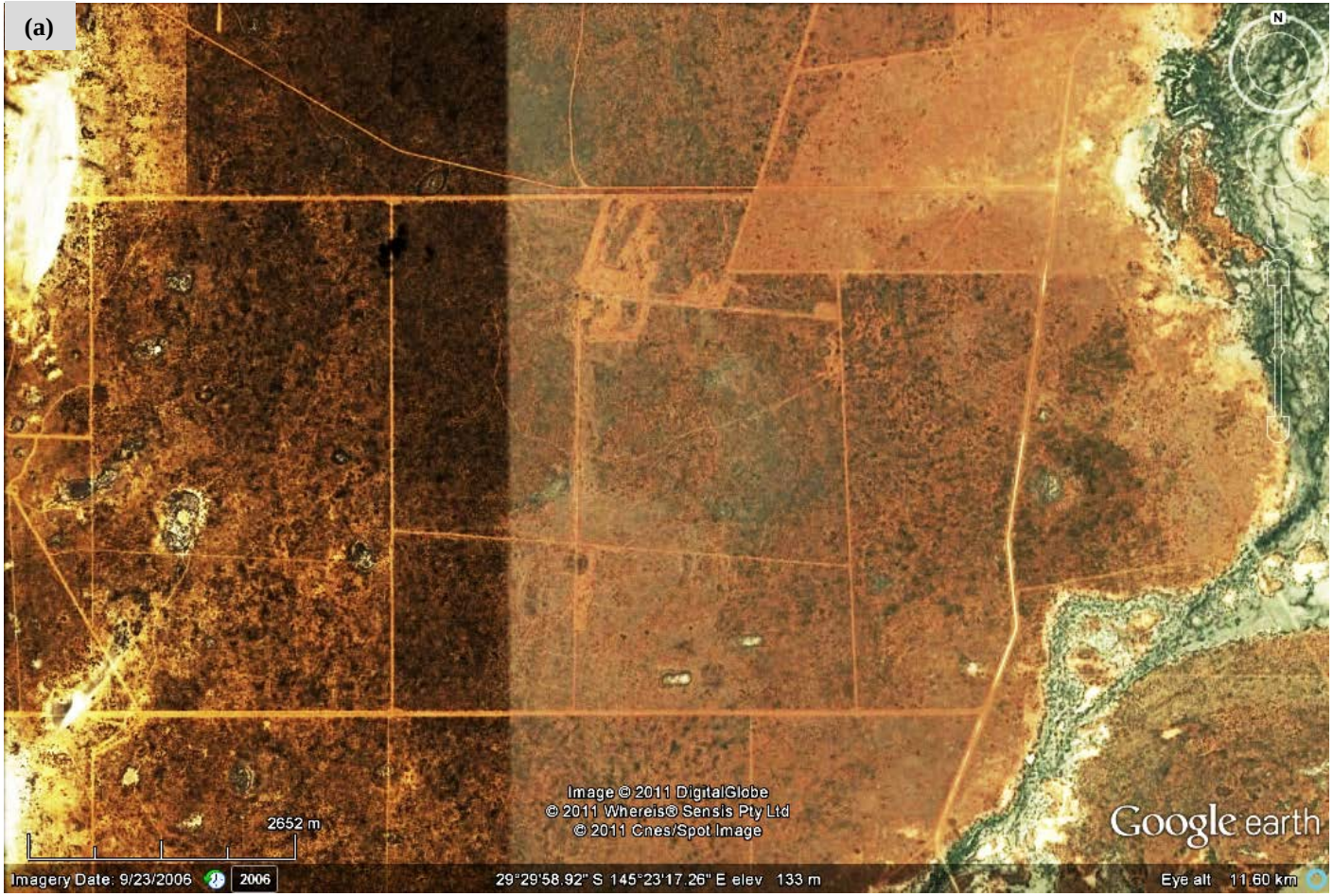
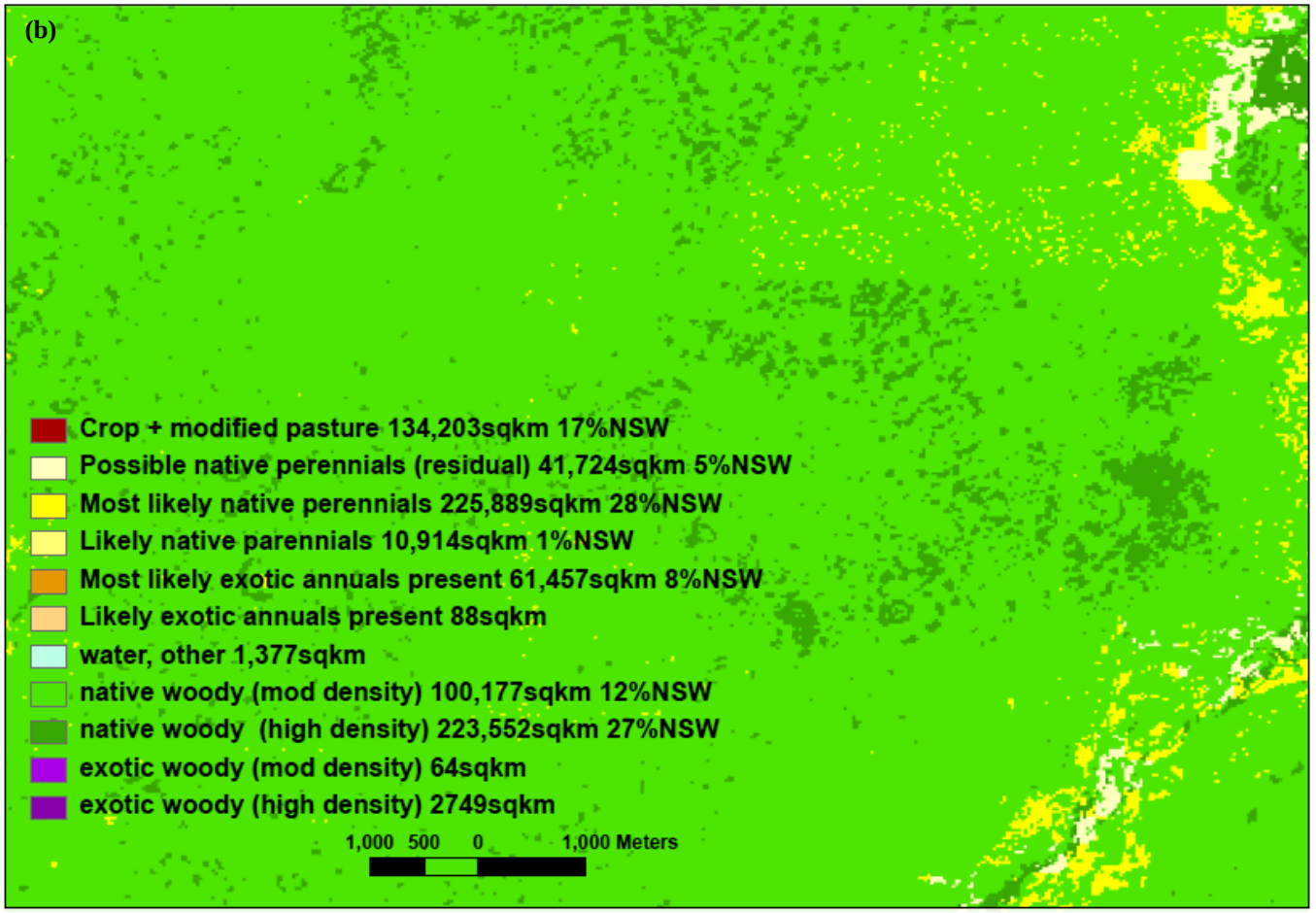
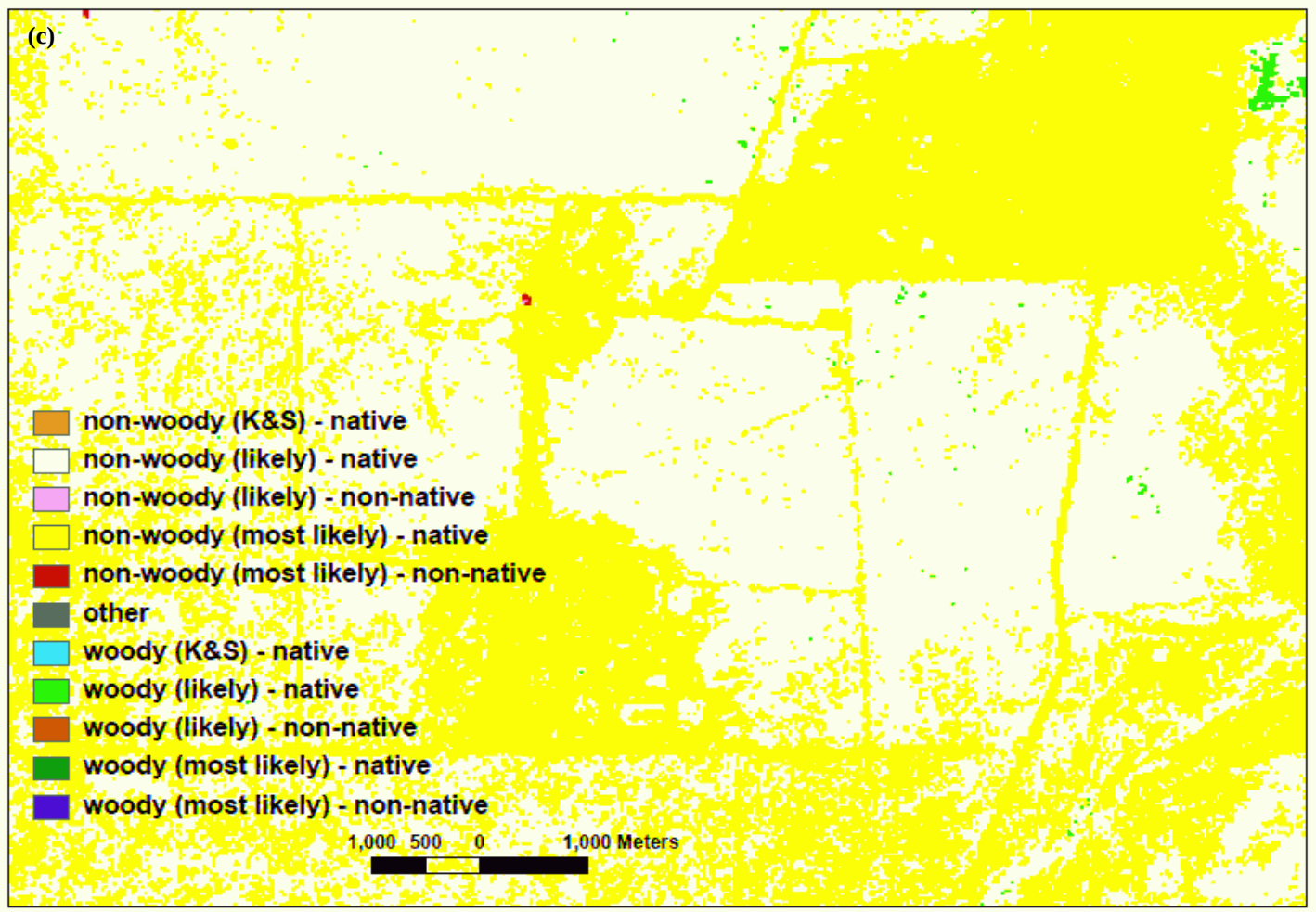


Figure A-V-2. Example of refinement of area of remnant native vegetation: city of Broken Hill **(a)** Supposedly extant native vegetation: many roads and ditches (dark green) were marked as native woody vegetation and the vast majority of land was marked as native perennials [native grasses, forbs and low shrubs etc.]. Consequently it was necessary to subtract anthropogenic alterations, as deduced from the land-use layer, **(b),** and to buffer the road areas, to leave more realistic, extant native vegetation: blue areas in **(c)**.

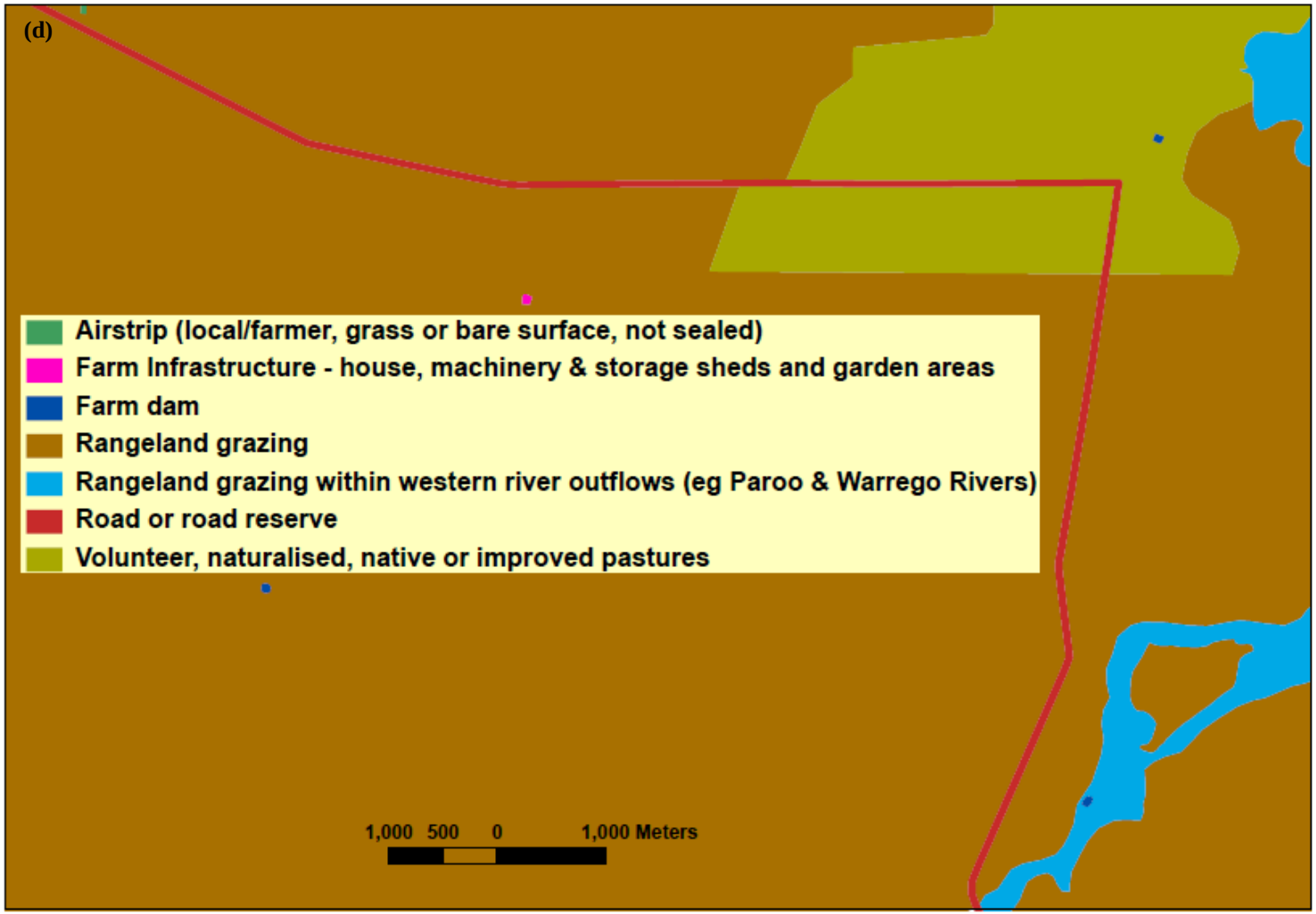

(a)
N
Image © 2011 DigitalGlobe
© 2011 Whereis® Sensis Pty Ltd
© 2011 Cnes/Spot Image
2652 m
Google earth
Imagery Date: 9/23/2006
2006
29°29'58.92" S 145°23'17.26" E elev 133 m
Eye alt 11.60 km


(b)
Crop + modified pasture 134,203sqkm 17%NSW
Possible native perennials (residual) 41,724sqkm 5%NSW
Most likely native perennials 225,889sqkm 28%NSW
Likely native parennials 10,914sqkm 1%NSW
Most likely exotic annuals present 61,457sqkm 8%NSW
Likely exotic annuals present 88sqkm
water, other 1,377sqkm
native woody (mod density) 100,177sqkm 12%NSW
native woody (high density) 223,552sqkm 27%NSW
exotic woody (mod density) 64sqkm
exotic woody (high density) 2749sqkm
1,000 500 0 1,000 Meters

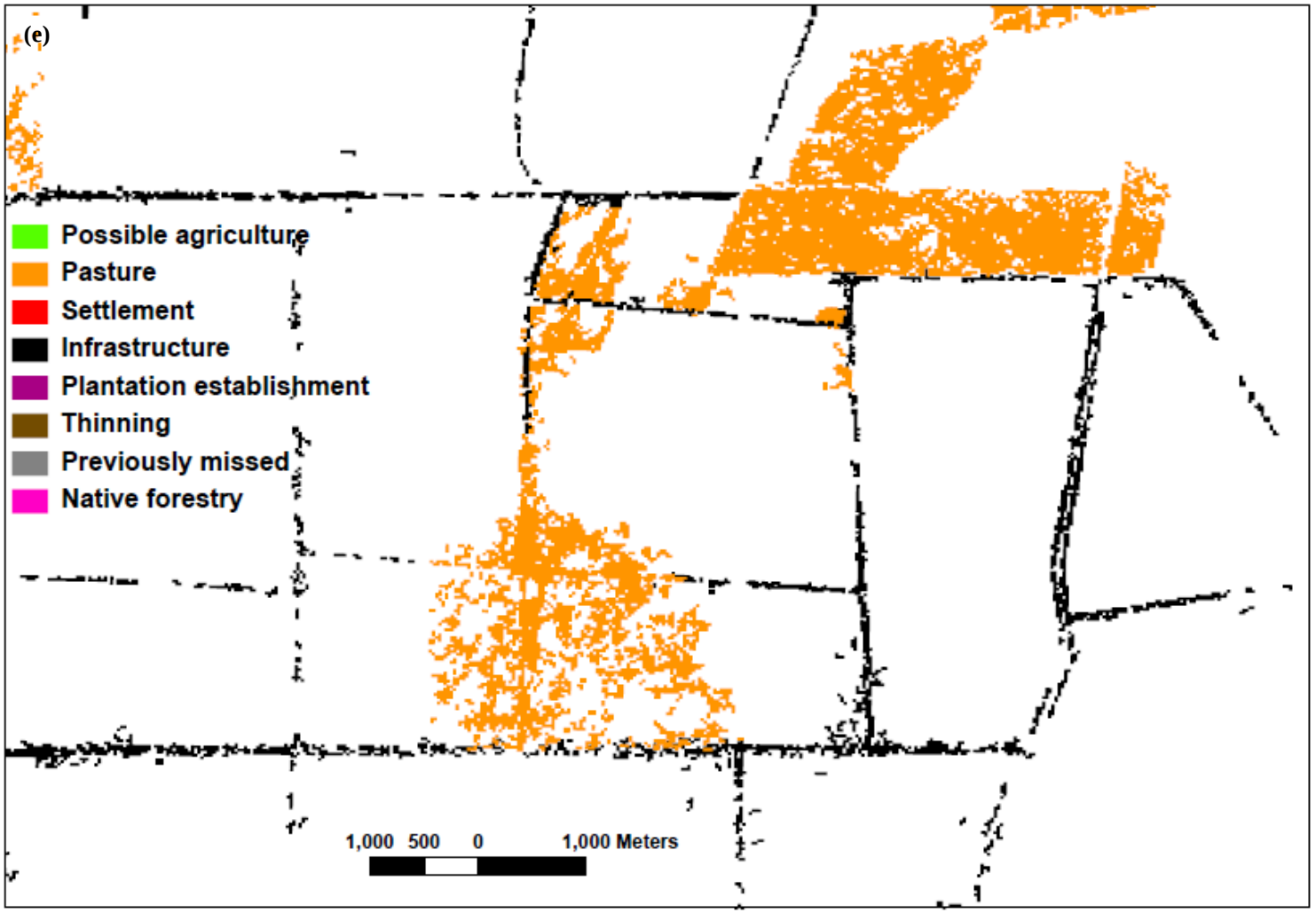
(b)


(c)
non-woody (K&S) - native
non-woody (likely) - native
non-woody (likely) - non-native
non-woody (most likely) - native
non-woody (most likely) - non-native
other
woody (K&S) - native
woody (likely) - native
woody (likely) - non-native
woody (most likely) - native
woody (most likely) - non-native
1,000 500 0 1,000 Meters

(c)


(d)
Airstrip (local/farmer, grass or bare surface, not sealed)
Farm Infrastructure - house, machinery & storage sheds and garden areas
Farm dam
Rangeland grazing
Rangeland grazing within western river outflows (eg Paroo & Warrego Rivers)
Road or road reserve
Volunteer, naturalised, native or improved pastures
1,000 500 0 1,000 Meters

(d)


(e)
Possible agriculture
Pasture
Settlement
Infrastructure
Plantation establishment
Thinning
Previously missed
Native forestry
1,000 500 0 1,000 Meters


(f)
Remnant native Mitchell Landscapes after buffering
Paroo-Warrego Channels & Floodouts
Paroo-Warrego Linear Dunes
Road_Arterial
Road_Major
Road_Minor
Road_VMinor
1,000 500 0 1,000 Meters

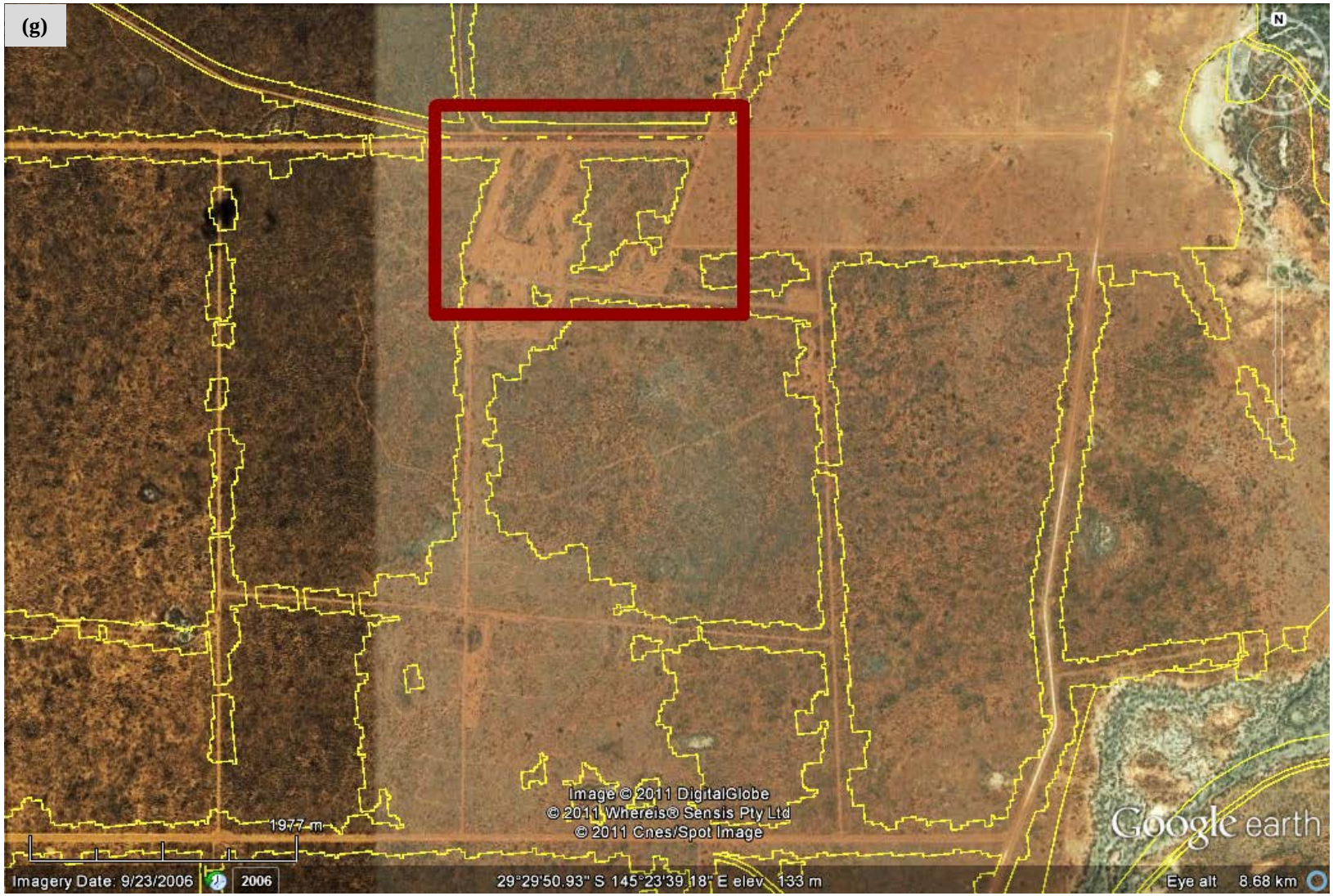
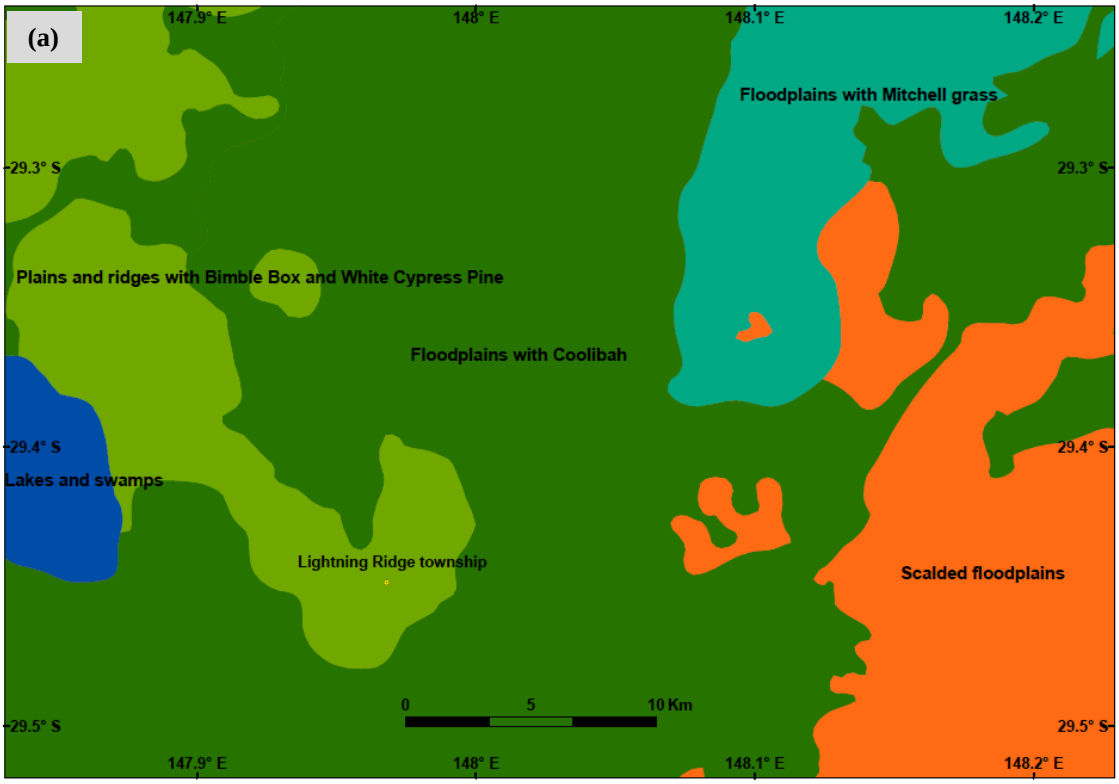
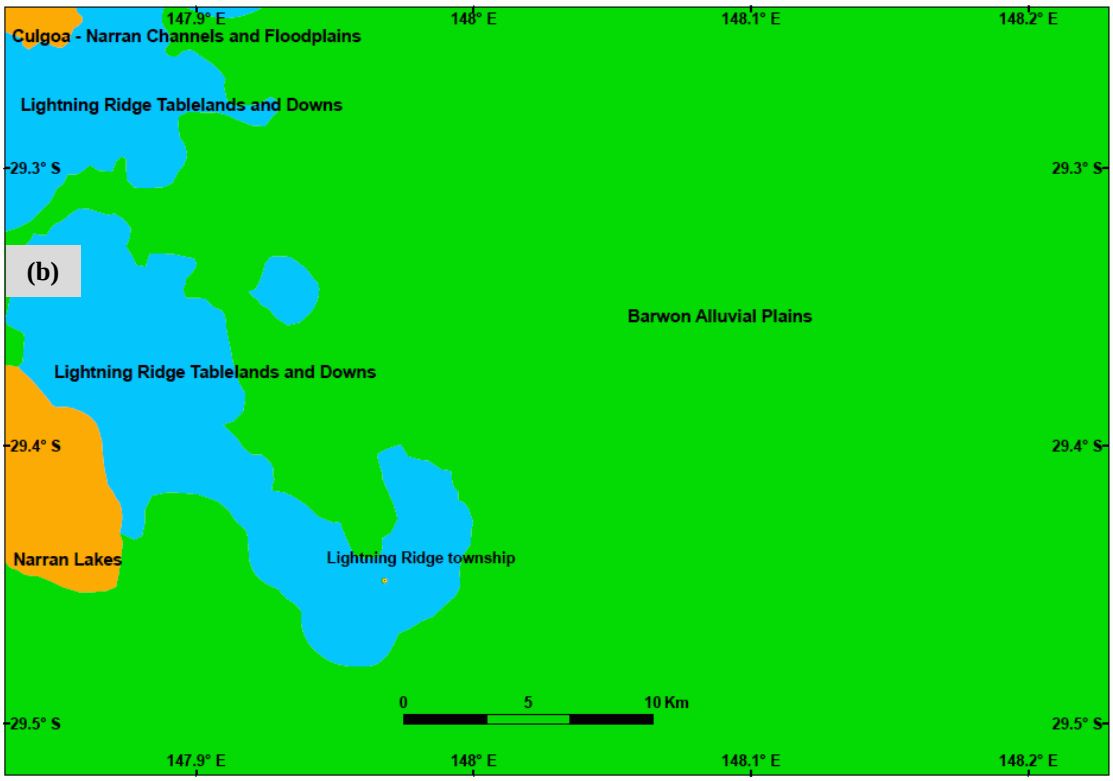
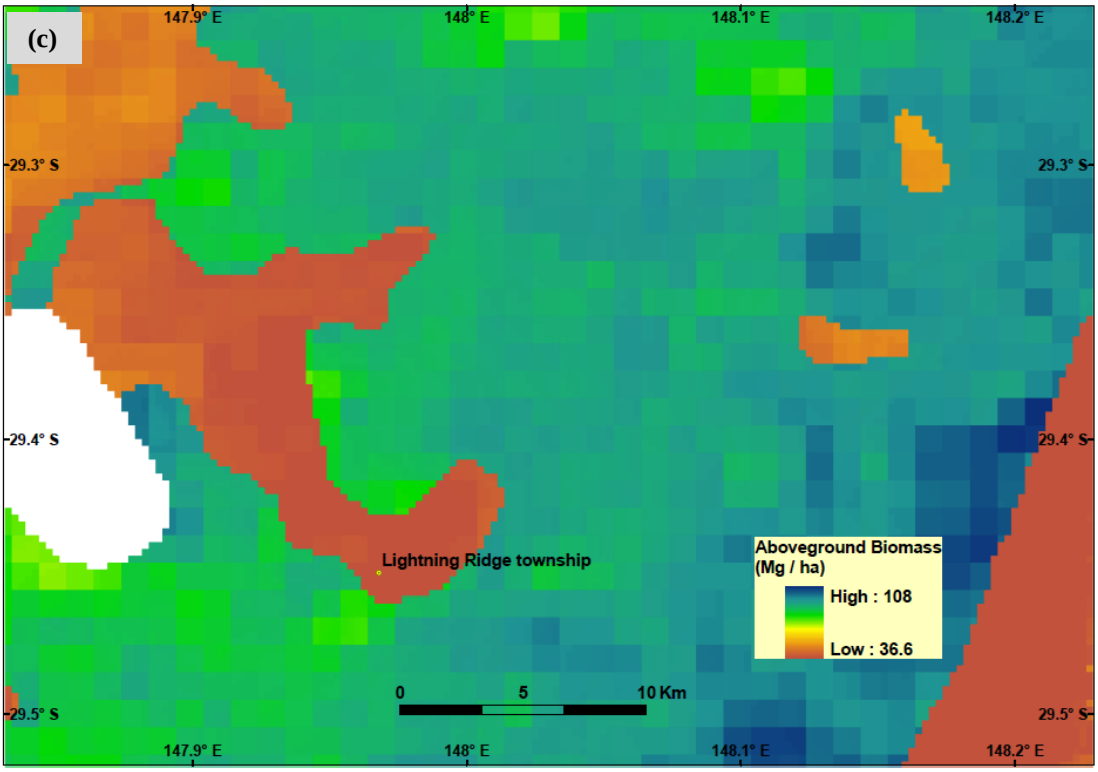
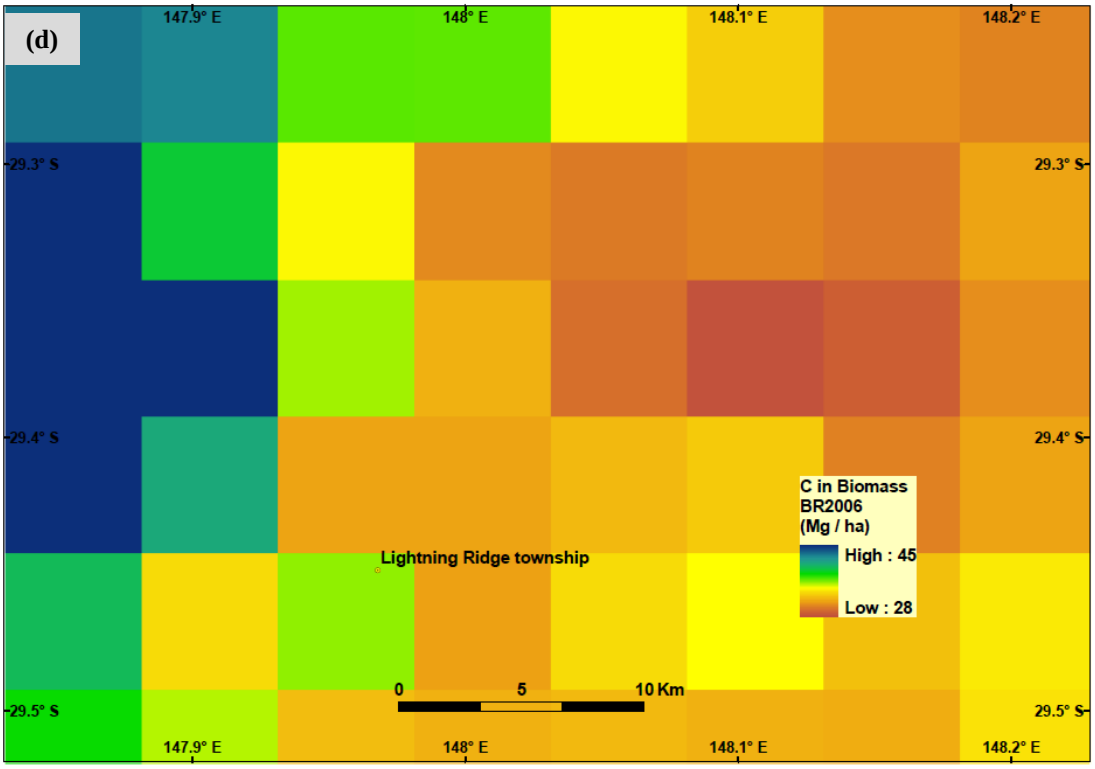

(g)
N
Image © 2011 DigitalGlobe
© 2011 Whereis® Sensis Pty Ltd
© 2011 Cnes/Spot Image
1977 m
Google earth
Imagery Date: 9/23/2006
2006
29°29'50.93" S 145°23'39.18" E elev 133 m
Eye alt 8.68 km

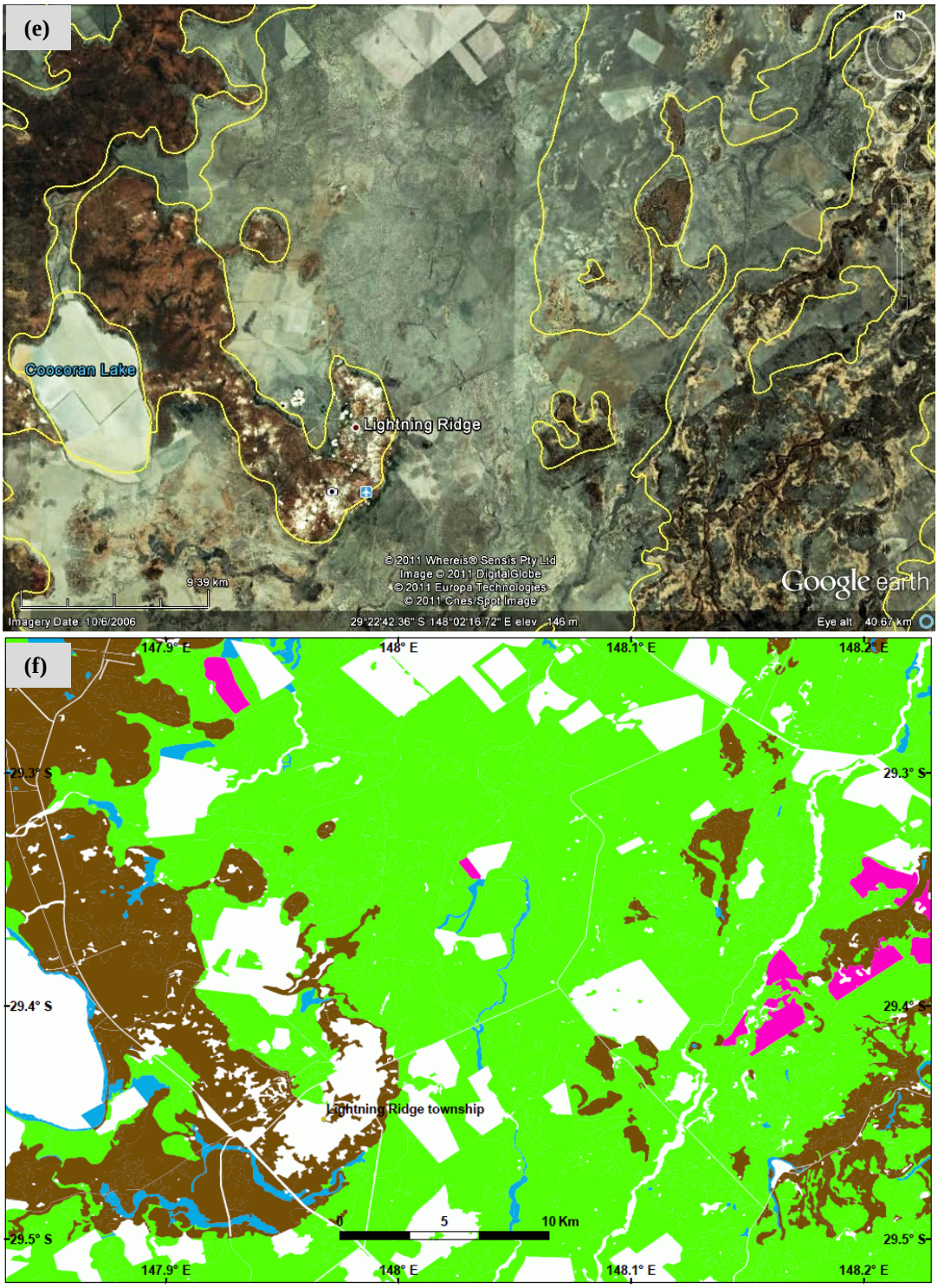
Figure A-V-3. Example refinement of areas of remnant ecosystems, factors considered in determining remnant vegetation within ecosystem-types. More detail corresponding to Figure 10-4.a. Site is ~86 km NW of Bourke, centred at Lat/Long -29.523°/145.406°, with main ecosystem-types ('Mitchell Landscapes) being Paroo-Warrego Mulga sandplains and dunefields, with the Warrego River floodplain in the SE corner. **(a)** GoogleEarth© Inc image of 23-Sep-2006 showing clearing, livestock tracks, vehicular tracks grazing land (deforested and non-deforested), roads and farm infrastructure. (The dark/light—left/right contrast is an imaging artefact.) **(b)** native vegetation extent map, showing deforested land incorrectly delineated as native woody or native perennials. **(c)** another native vegetation extent map showing incorrect listing of most of the area as likely non-woody, or most like native. **(d)** land-use map showing discrimination of major deforestation and major roads. **(e)** deforestation map (from LANDSAT 1988–2007), showing loss of trees within paddocks and along some road lines, and classifications of land-use upon deforestation. **(f)** The ecosystem-types after extraction of disturbance buffers to leave 'remnant' vegetation. **(g)** The derived remnant vegetation polygons (overlaid on GoogleEarth© image) containing predominantly vestigial forest with attrition due to livestock and vehicular tracks, and unmapped roads. red box is area of Figure 10-4.a.


(a)
147.9° E
148° E
148.1° E
148.2° E
29.3° S
29.4° S
29.5° S
Floodplains with Mitchell grass
Plains and ridges with Bimble Box and White Cypress Pine
Floodplains with Coolibah
Lakes and swamps
Lightning Ridge township
Scalded floodplains
0
5
10 Km


(b)
147.9° E
148° E
148.1° E
148.2° E
29.3° S
29.4° S
29.5° S
Culgoa - Narran Channels and Floodplains
Lightning Ridge Tablelands and Downs
Barwon Alluvial Plains
Lightning Ridge Tablelands and Downs
Narran Lakes
Lightning Ridge township
0
5
10 Km


(c)
147.9° E
148° E
148.1° E
148.2° E
29.3° S
29.4° S
29.5° S
Lightning Ridge township
Aboveground Biomass
(Mg / ha)
High : 108
Low : 36.6
0
5
10 Km


(d)
147.9° E
148° E
148.1° E
148.2° E
29.3° S
29.4° S
29.5° S
Lightning Ridge township
C in Biomass
BR2006
(Mg / ha)
High : 45
Low : 28
0
5
10 Km


(e)
Coocoran Lake
Lightning Ridge
9.39 km
© 2011 Whereis® Sensis Pty Ltd
Image © 2011 DigitalGlobe
© 2011 Europa Technologies
© 2011 Cnes/Spot Image
Google earth
Imagery Date: 10/6/2006
29°22'42.36" S 148°02'16.72" E elev 146 m
Eye alt 40.67 km
(f)
147.9° E
148° E
148.1° E
148.2° E
29.3° S
29.4° S
29.5° S
Lightning Ridge township
0
5
10 Km

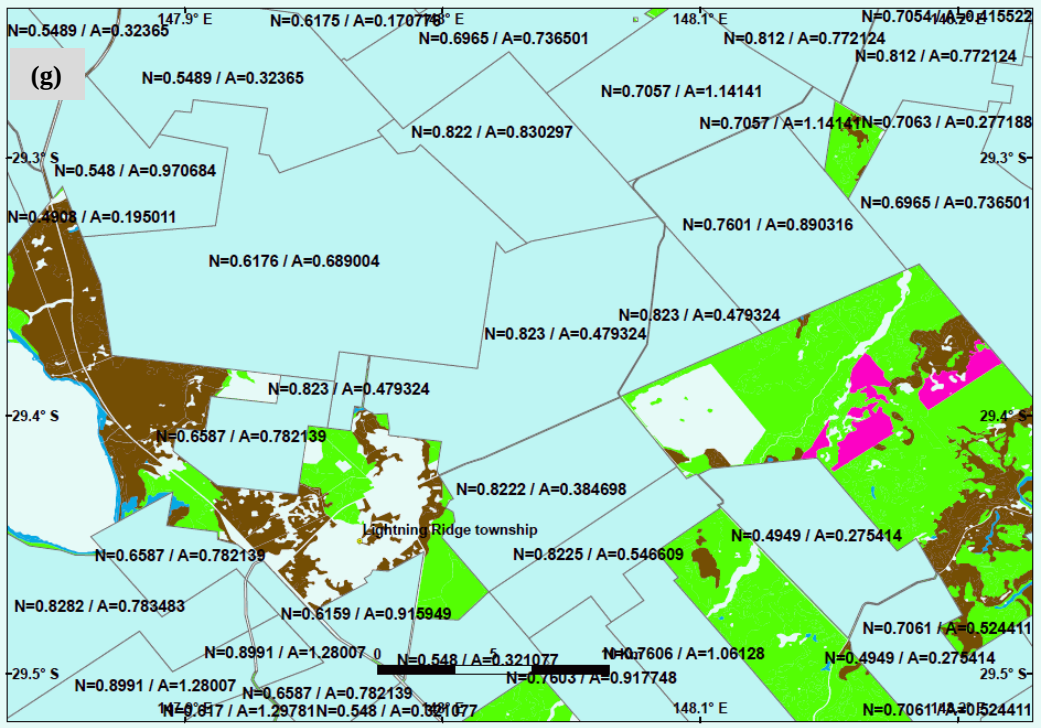
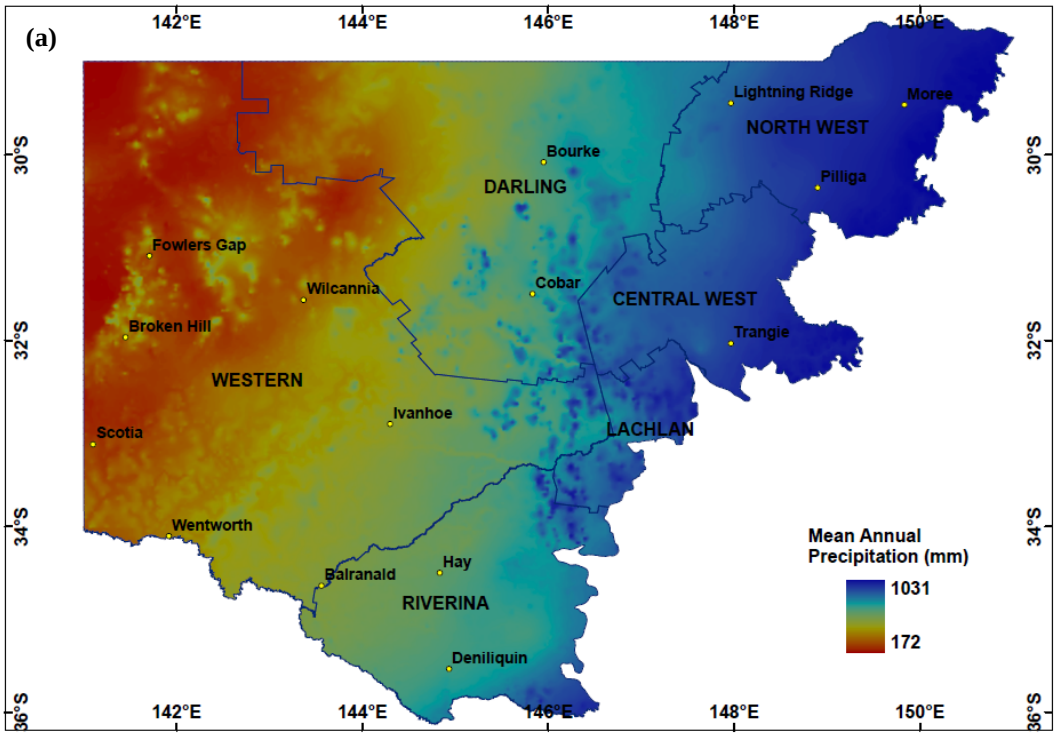
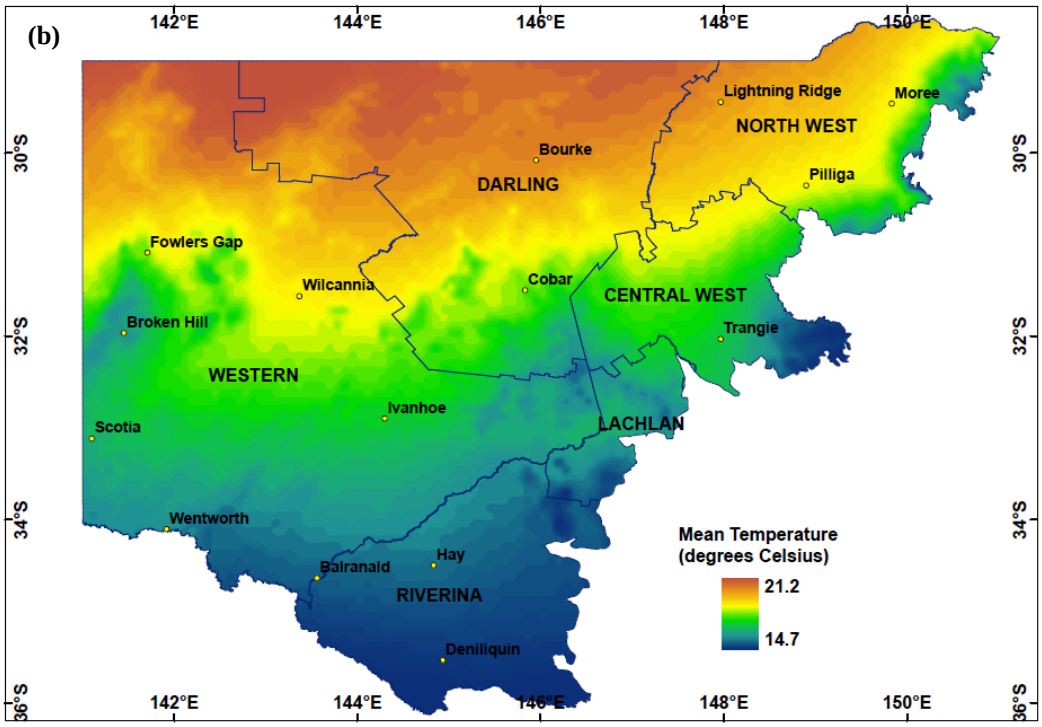


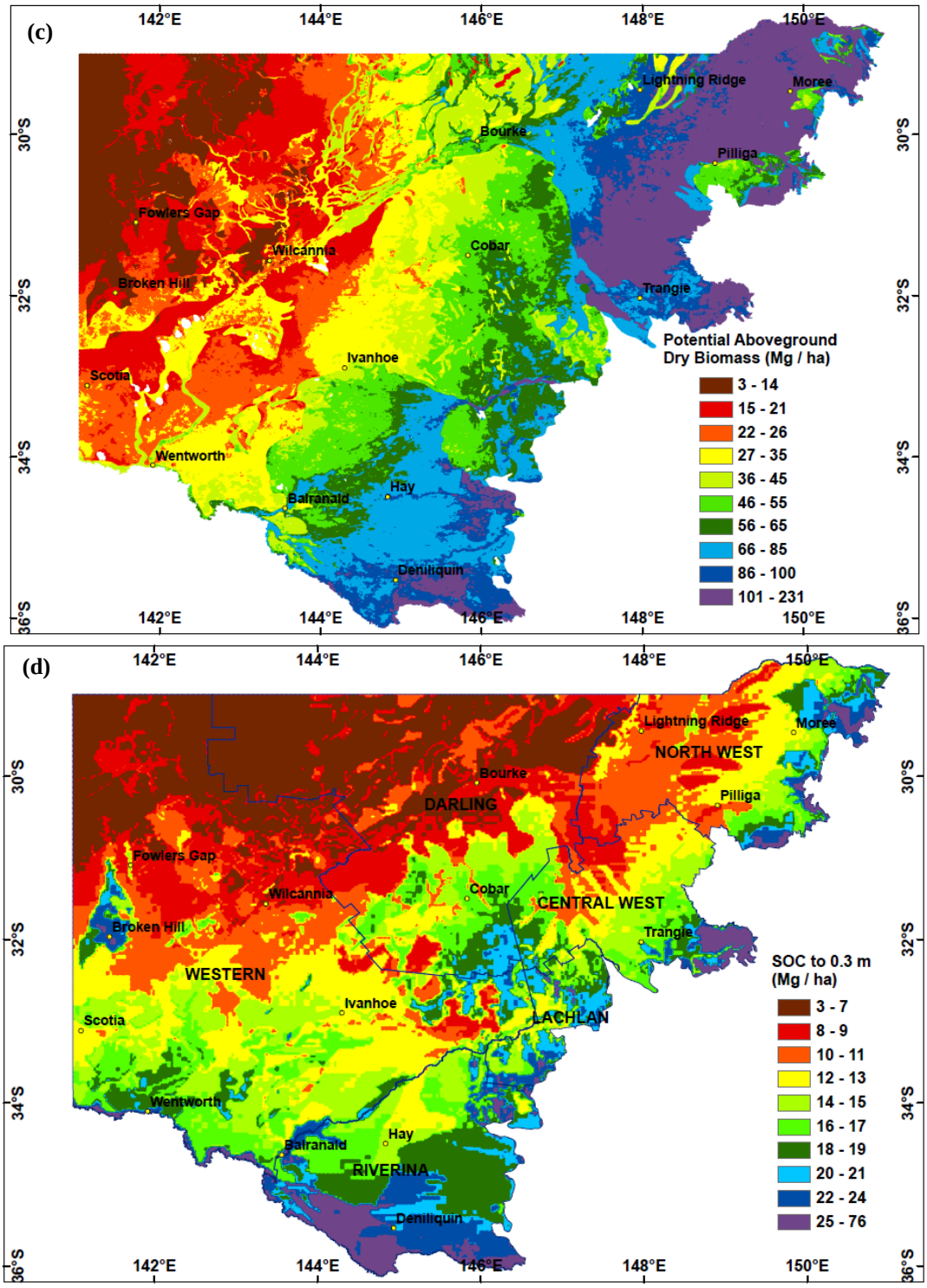
Figure A-V-4. Different resolution of land system definitions (**a**, **e**). Western-NSW-only data and (b) ecosystem-type data. NCAS potential aboveground biomass **(c)** showing reliance on soil mapping, but not recognising absence of woody vegetation in native grasslands. **(d)** BR2006 potential biomass showing recognition of differences in woody cover (e.g. 'Bimble Box and White Cypress Pine' from 'scalded plains' and 'Mitchell Grass plains'), though with ~5 km pixels it was too coarse to discriminate some paddock boundaries and small lakes (GoogleEarth Image **(e)**). Individual 'properties' **(g)** may overlap more than one land use **(f)**, including different types of rangeland and rangeland-plus-cropland— the lighter-coloured paddocks in (e). Land use colour scheme in (f) and (g) same as Figure A-V-2. Carrying capacities in (g) are notional 'N' over actual (extant) 'A' in dse ha^{-1}.

The potential biomass NCAS data (~250 m pixels), was calculated using a formula that relying on soil characteristics with ~1 km resolution. Also, it does not take into account where native grassland prevails, and assigns them too high a biomass, based on trees coverage. Such grasslands, often on alluvial clays, have lower biomass than the surrounding savannah.

Only datasets covering the whole study region were used. Some ecosystem data were available at higher resolution, for limited areas **(a)**. There were about 0.59 Mha of grasslands thus delineated, which in the ecosystem-type layer, were merged with surrounding savannah, scalded plains and floodplains. That grassland was not all rangeland and would have contributed no more than ~1% of the rangeland area and therefore it was considered acceptable to use only the ecosystem-type data.


(a)
142°E
144°E
146°E
148°E
150°E
30°S
32°S
34°S
36°S
Lightning Ridge
Moree
NORTH WEST
Bourke
DARLING
Pilliga
Fowlers Gap
Wilcannia
Cobar
CENTRAL WEST
Broken Hill
Trangie
WESTERN
Ivanhoe
LACHLAN
Scotia
Wentworth
Hay
Balranald
RIVERINA
Deniliquin
Mean Annual
Precipitation (mm)
1031
172


(b)
142°E
144°E
146°E
148°E
150°E
30°S
32°S
34°S
36°S
Lightning Ridge
Moree
NORTH WEST
Bourke
DARLING
Pilliga
Fowlers Gap
Wilcannia
Cobar
CENTRAL WEST
Broken Hill
Trangie
WESTERN
Ivanhoe
LACHLAN
Scotia
Wentworth
Hay
Balranald
RIVERINA
Deniliquin
Mean Temperature
(degrees Celsius)
21.2
14.7


(c)
142°E
144°E
146°E
148°E
150°E
30°S
32°S
34°S
36°S
Lightning Ridge
Moree
Bourke
Pilliga
Fowlers Gap
Wilcannia
Cobar
Broken Hill
Trangie
Ivanhoe
Scotia
Wentworth
Hay
Balranald
Deniliquin
Potential Aboveground Dry Biomass (Mg / ha)
3 - 14
15 - 21
22 - 26
27 - 35
36 - 45
46 - 55
56 - 65
66 - 85
86 - 100
101 - 231
(d)
NORTH WEST
DARLING
CENTRAL WEST
WESTERN
LACHLAN
RIVERINA
SOC to 0.3 m (Mg / ha)
3 - 7
8 - 9
10 - 11
12 - 13
14 - 15
16 - 17
18 - 19
20 - 21
22 - 24
25 - 76

(d)

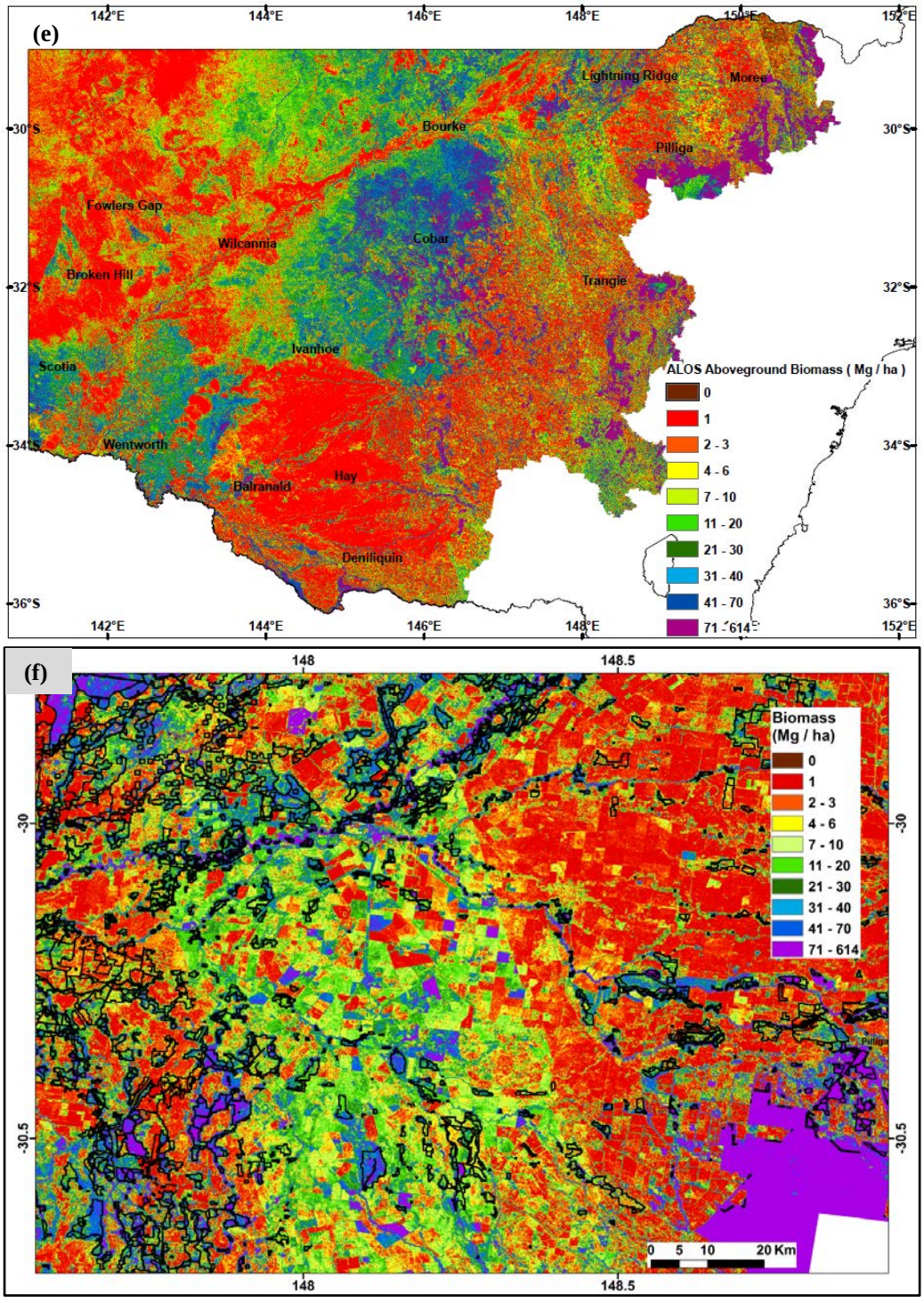


Figure A-V-5. Maps of key attributes for the rangeland zone of NSW. Temperature and SOC roughly follow a north-south trend; rainfall and biomass roughly follow a west-east trend. (e) measured aboveground biomass as of 2009 from ALOS imagery. The striping in the higher-rainfall region of arable land in the east, which had negligible influence on the rangeland biomass benchmarks (black outlines in (f)). **(f)** is the most prominently striping in (e), 50 km west of Pilliga.

The striping of ALOS PALSAR biomass between neighbouring, ~55 km wide swaths ((e), (f)) was investigated to see if it had contributed to error margins. Stripe edges could yield the greatest error in local carbon flux. Within a stripe, deduction of which land was below par would still remain correct (as it is relative), but the magnitude of change would be incorrect. Such striping has been attributed to different soil or vegetation moisture on different image acquisition dates and mis-correction for that during mosaicking (Armston et al., 2010; Lucas et al., 2010). There were 1.9 Mha (1,895,593 ha) or 6.2% of the NSW total commercial rangelands in the vicinity of the striping and therefore potentially affected, which was around Lightning Ridge, across to Bourke; a small region north of Pilliga; and some near Trangie. Land types potentially affected were alluvial plains, channels and floodplains of the Darling River. The percentage of freehold land in the area was high, at 28% (0.54 Mha), which was 21% of the total NSW freehold. Most of the potentially affected area (i.e. 90%) had an extant biomass of 31 Mg ha^{-1} (as per the ALOS data) but 50% had an extant biomass of only 25 Mg ha^{-1}. However, the striping was only manifested under fine spectral resolution for low-end biomass: the edge contrast being between 1 and 3–9 Mg ha^{-1}, and it was indiscernible between pixels with higher biomass. The latter fact suggests it was a soil or leaf moisture effect and that the rain had not yet influenced woody biomass. On closer examination the striping appeared to be almost exclusively in arable land (possibly due to minimal ground cover, including leaf litter), and not in commercial rangeland. Thus the rangeland areas considered above as being potentially affected were most likely unaffected and the effect on carbon calculations was negligible.

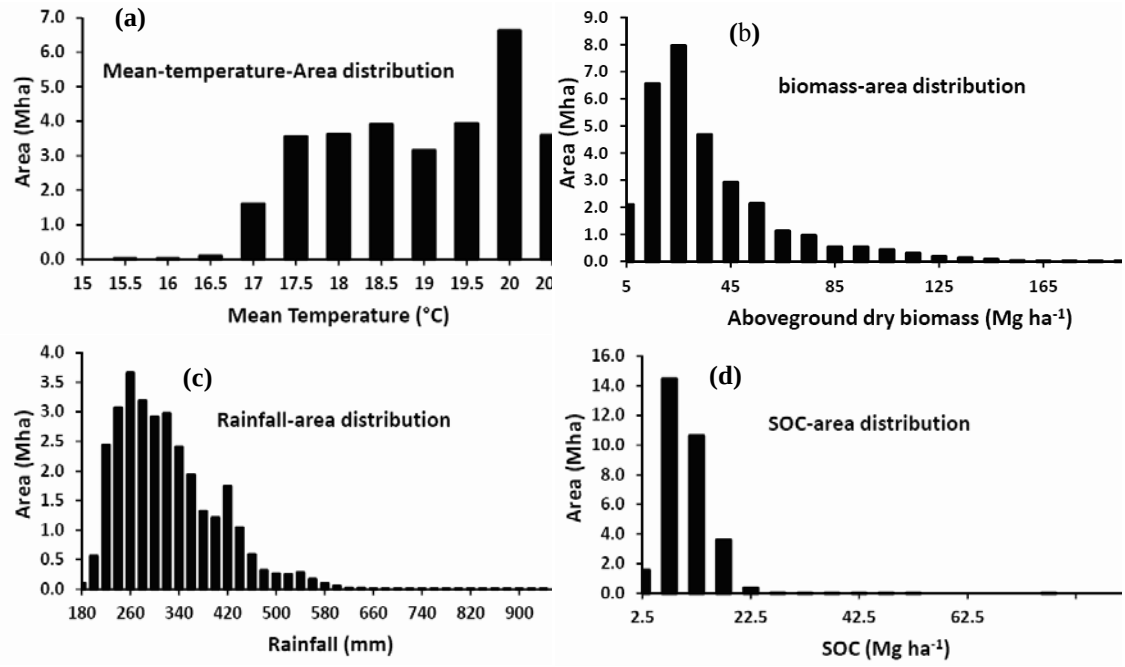
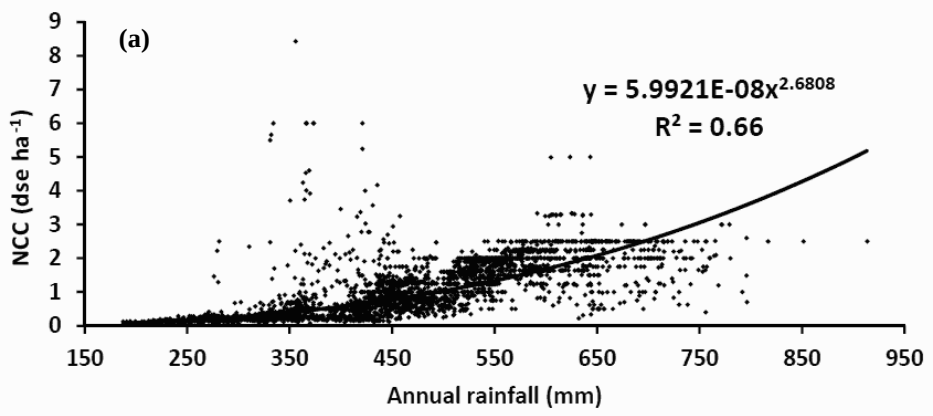


Figure A-V-6. Distribution histograms for attributes of NSW commercial rangelands. Note that all are strongly negatively skewed except for mean annual temperature, which is positively skewed. Note that **(b)** is potential aboveground dry biomass (from the NCAS layer), and not actual biomass. Biomass, rainfall and SOC all showed strong positive skew, i.e. more concentrated towards low values. That skew is probably due to the small amount of land with higher rainfall on the eastern edge of the rangeland zone and lower temperature in the south-eastern corner. Temperature showed the opposite trend, with a strong negative skew, although again owing to that SE corner.



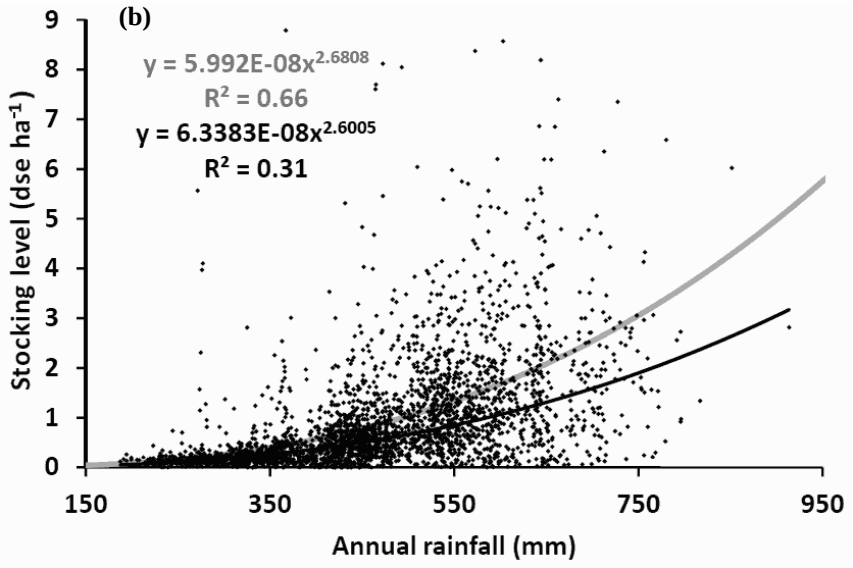
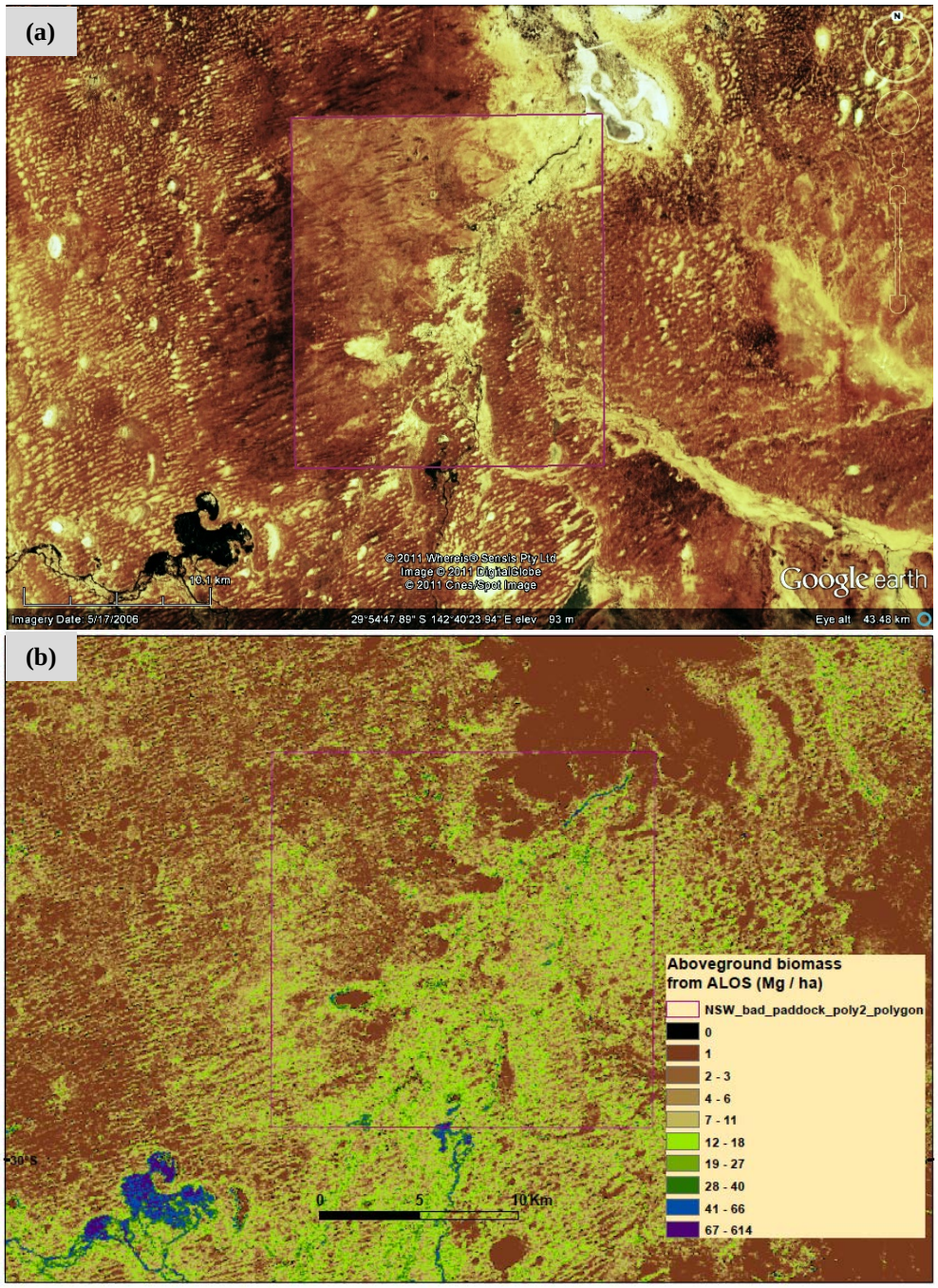


Figure A-V-7. Stocking levels versus rainfall. This relationship was the most significant out of those tested (potential biomass, SOC and rainfall). **(a)** Notional Carrying Capacity, and **(b)** actual stocking level, with notional (in grey) overlaid for comparison.


(a)
© 2011 WhereIs© Sensis Pty Ltd
Image © 2011 DigitalGlobe
© 2011 Cnes/Spot Image
Google earth
10.1 km
Imagery Date: 5/17/2006
29°54'47.89" S 142°40'23.94" E elev 93 m
Eye alt 43.48 km
(b)
Aboveground biomass
from ALOS (Mg / ha)
NSW_bad_paddock_poly2_polygon
0
1
2 - 3
4 - 6
7 - 11
12 - 18
19 - 27
28 - 40
41 - 66
67 - 614
30°S
0
5
10 Km

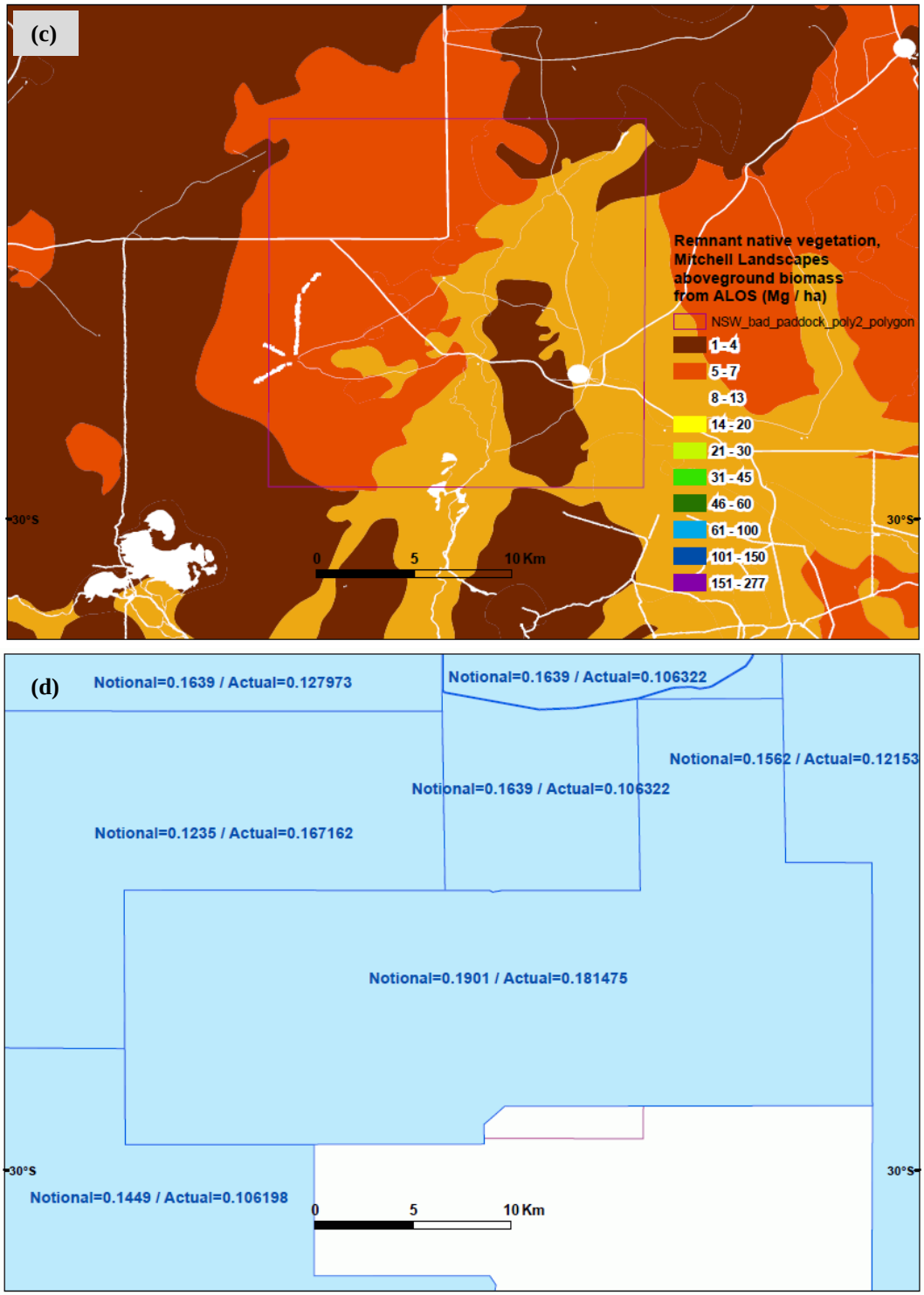
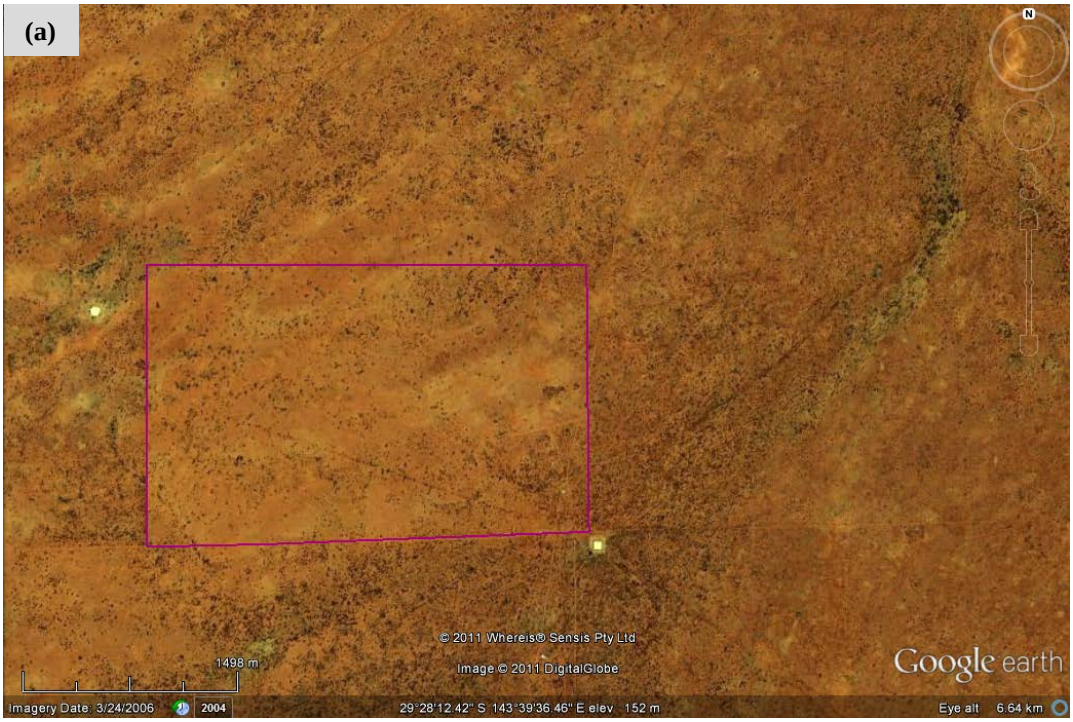


Figure A-V-8. Example vegetation-cover contrast due to land-use, land-use GIS layer attributes and ecosystem-types. **(a)** GoogleEarth image showing contrast across a diagonal line (from the centre towards the top-left of the image). **(b)** The contrast is reflected in the ALOS biomass layer. Most of the area is classified as unmodified native vegetation with a land-use of rangeland, and therefore it is

'remnant' native vegetation, which was used in obtaining values for carbon benchmarks of ecosystem-types (Mitchell-Landscapes). Note that the roads and riparian areas were buffered and subtracted and not used in that calculation **(c)**. Properties from the LHPA livestock census do not match the land-use pattern observed in the vegetation **(d)** (e.g. the diagonal line showing contrast in (a) and (b) does not correspond to a property boundary in (d)). Therefore one cannot expect correlation between land-use according to actual livestock stocking levels, and detected differences in vegetation biomass. Note that if true 'remnant' vegetation had been delineated then higher values of biomass would be obtained for representative pre-commercialisation vegetation (useful as benchmarks).



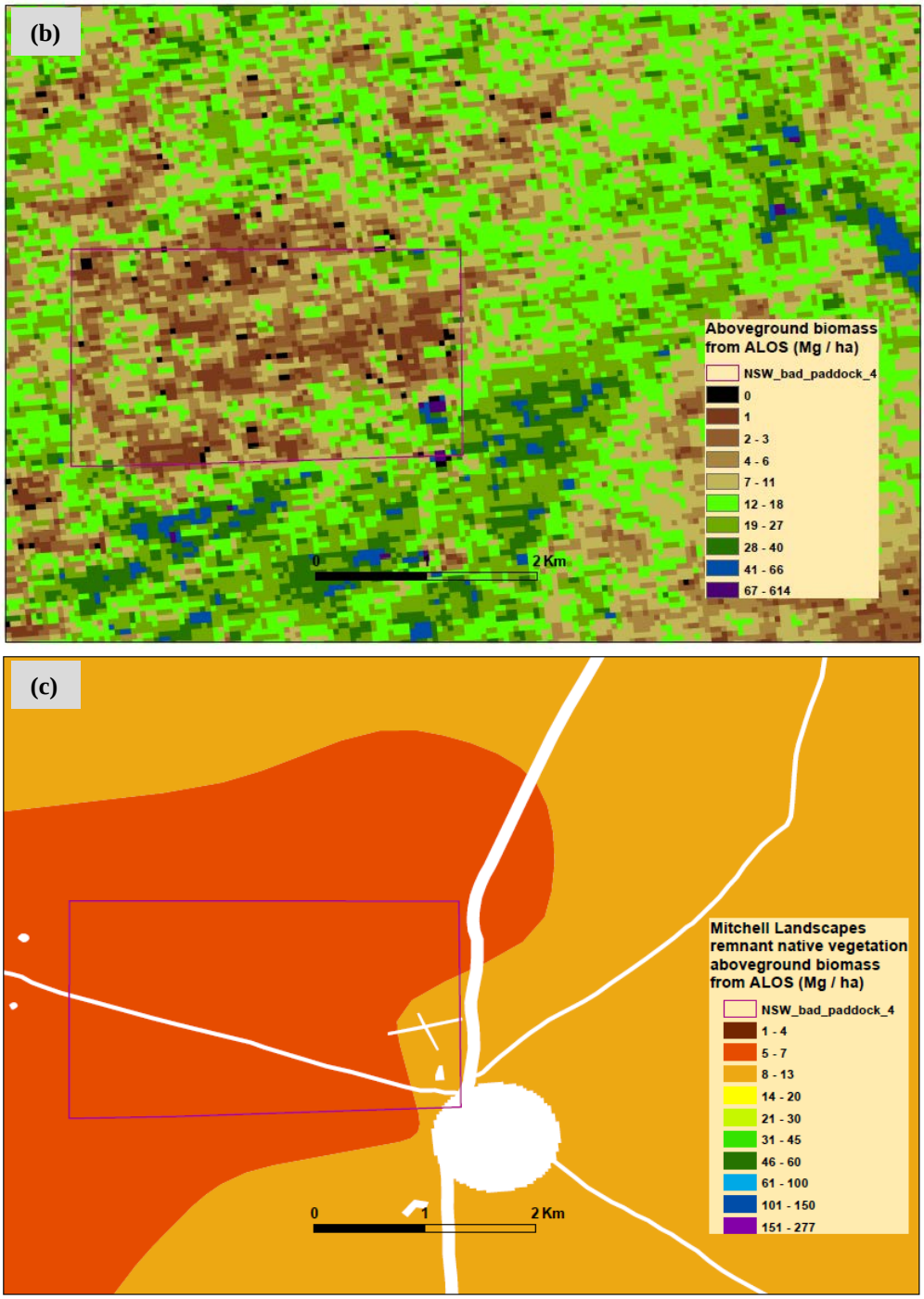


Figure A-V-9. Example vegetation-cover contrast due to land-use showing in GoogleEarth **(a)** and in the ALOS biomass layer **(b)**. Land-use GIS layer attributes and ecosystem-types (Mitchell-Landscapes) showing buffering of roads and infrastructure, to leave 'remnant' native vegetation (although with obvious land-use attrition in some parts— within the pink rectangle) **(c)**. If true 'remnant' vegetation had been delineated then higher values of biomass would be obtained for benchmarks.

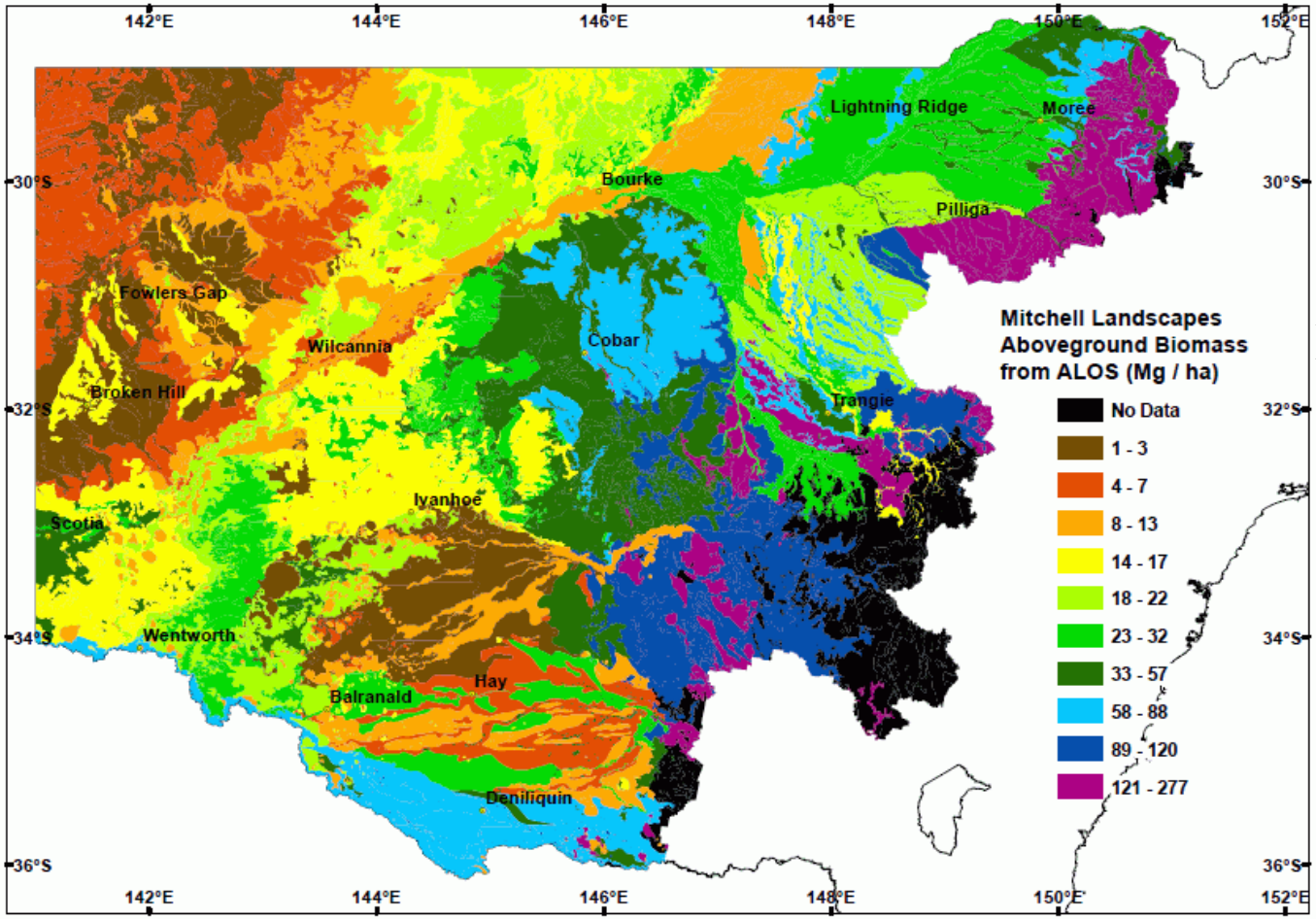


Figure A-V-10. Representative aboveground biomass for the ecosystem-types '(Mitchell Landscapes') as derived from remnant native vegetation and ALOS aboveground biomass scaled up to whole ecosystem-type.

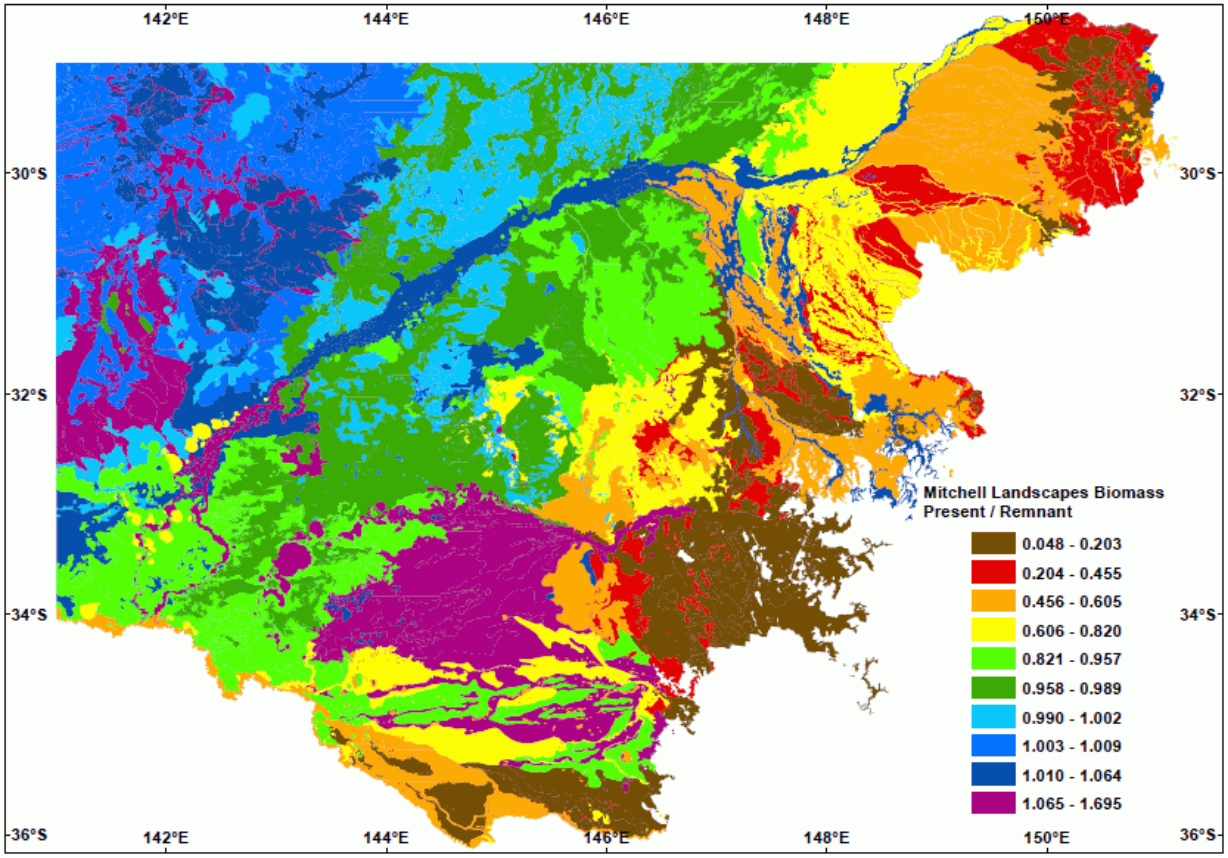


Figure A-V-11. Aboveground biomass ratios for ecosystem-types ('Mitchell Landscapes'). Value= extant average biomass / remnant's biomass. Lower values in east are most likely due to preferential deforestation of more-productive land. High values in the NW are probably due to the 'remnant' being degraded, i.e. artefacts of assessment. Ecosystem types with values > 1 (riparian and central floodplain areas), mostly likely due to riparian pixels not being considered as remnant (considered water-contaminated and vegetation unassigned, though confirmed as vegetation in GoogleEarth imagery), but they were included in the calculation of extant values.

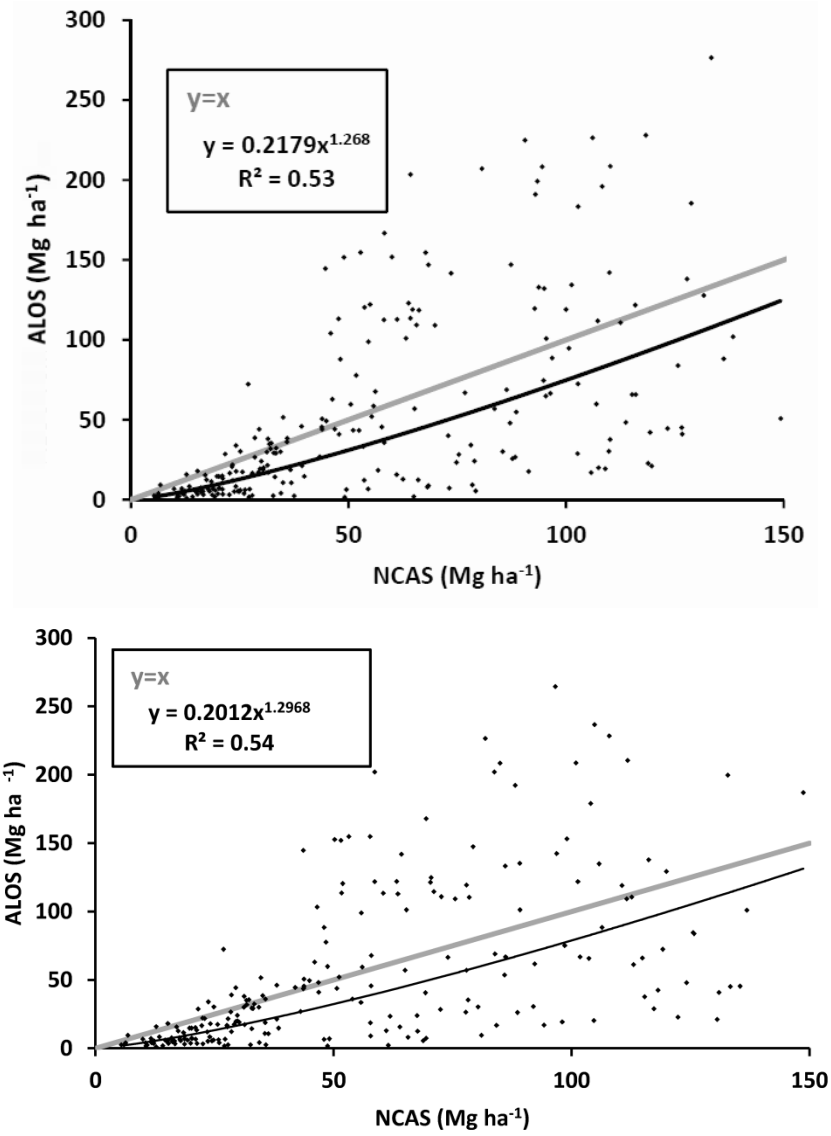


Figure A-V-12. ALOS aboveground biomass of remnant ecosystem-types versus NCAS potential biomass for: **(a)** whole ecosystem-types, and: **(b)** remnant vegetation in the ecosystem-types. The line of best fit lies below the 1:1 line, as would be expected due to degradation of remnant vegetation and earlier deforestation of the more productive land (i.e. higher biomass land). The average remnant ALOS value is only 14% and 12% below the NCAS potential for calculations (a) and (b) respectively.

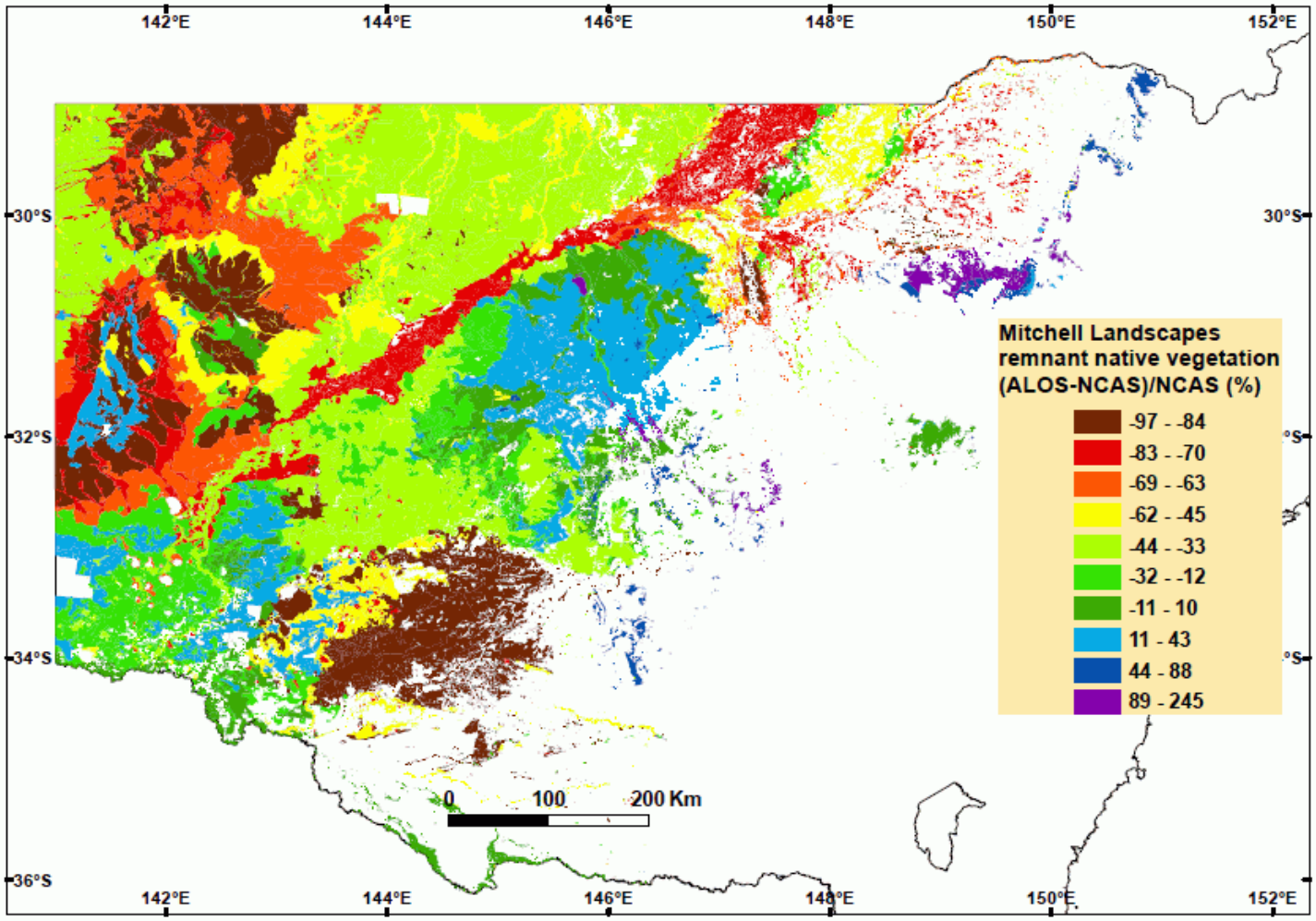


Figure A-V-13. Aboveground biomass of remnant ecosystem-types: ALOS compared with NCAS. ALOS values are notably lower on alluvial, riparian and grassland sites. NCAS values are notably lower than ALOS values in the higher-rainfall, taller forest areas. Explanations in Figure A-V-11, 12 captions.

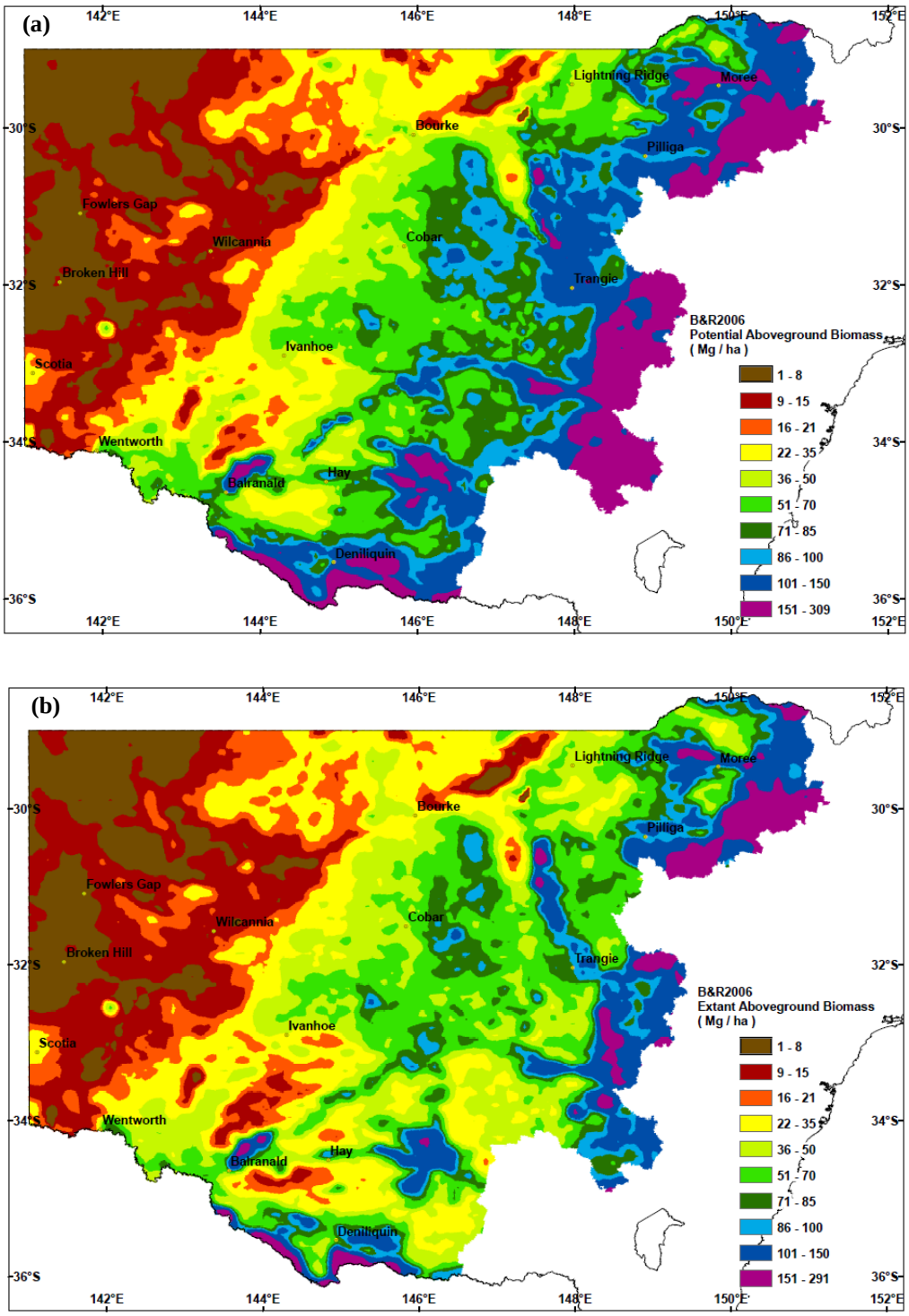
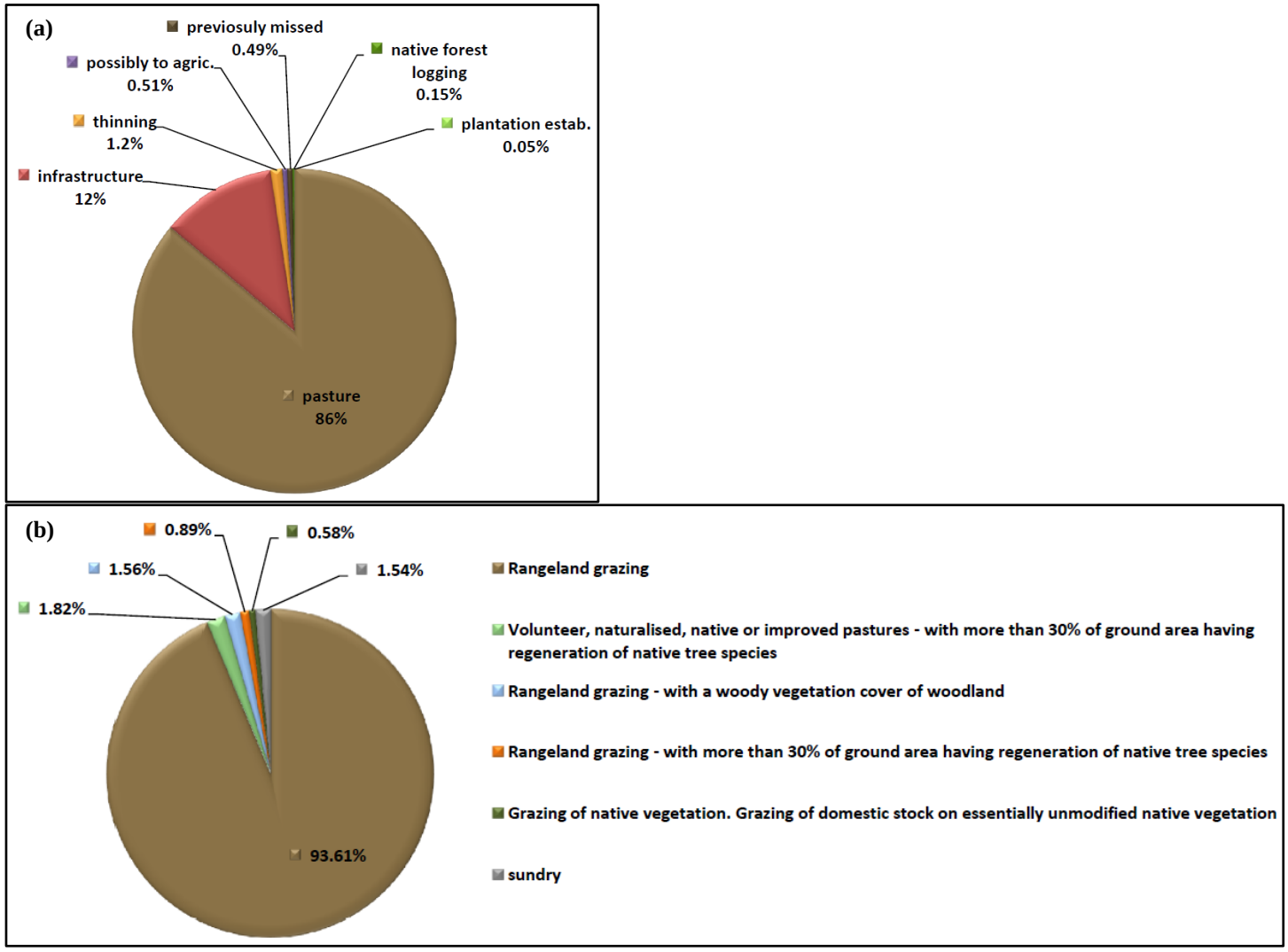


Figure A-V-14. Aboveground biomass for the six western-most LHPA districts, from the B&R2006 layer: **(a)** pre-commercialisation and **(b)** extant (Rangeland is a subset of this area.) Note highest losses in the east and slight gains in the 9-15 Mg ha^{-1} category in the west.


(a)
previosuly missed
0.49%
native forest
logging
0.15%
possibly to agric.
0.51%
thinning
1.2%
plantation estab.
0.05%
infrastructure
12%
pasture
86%
(b)
0.89%
0.58%
1.56%
1.54%
1.82%
93.61%
Rangeland grazing
Volunteer, naturalised, native or improved pastures - with more than 30% of ground area having regeneration of native tree species
Rangeland grazing - with a woody vegetation cover of woodland
Rangeland grazing - with more than 30% of ground area having regeneration of native tree species
Grazing of native vegetation. Grazing of domestic stock on essentially unmodified native vegetation
sundry

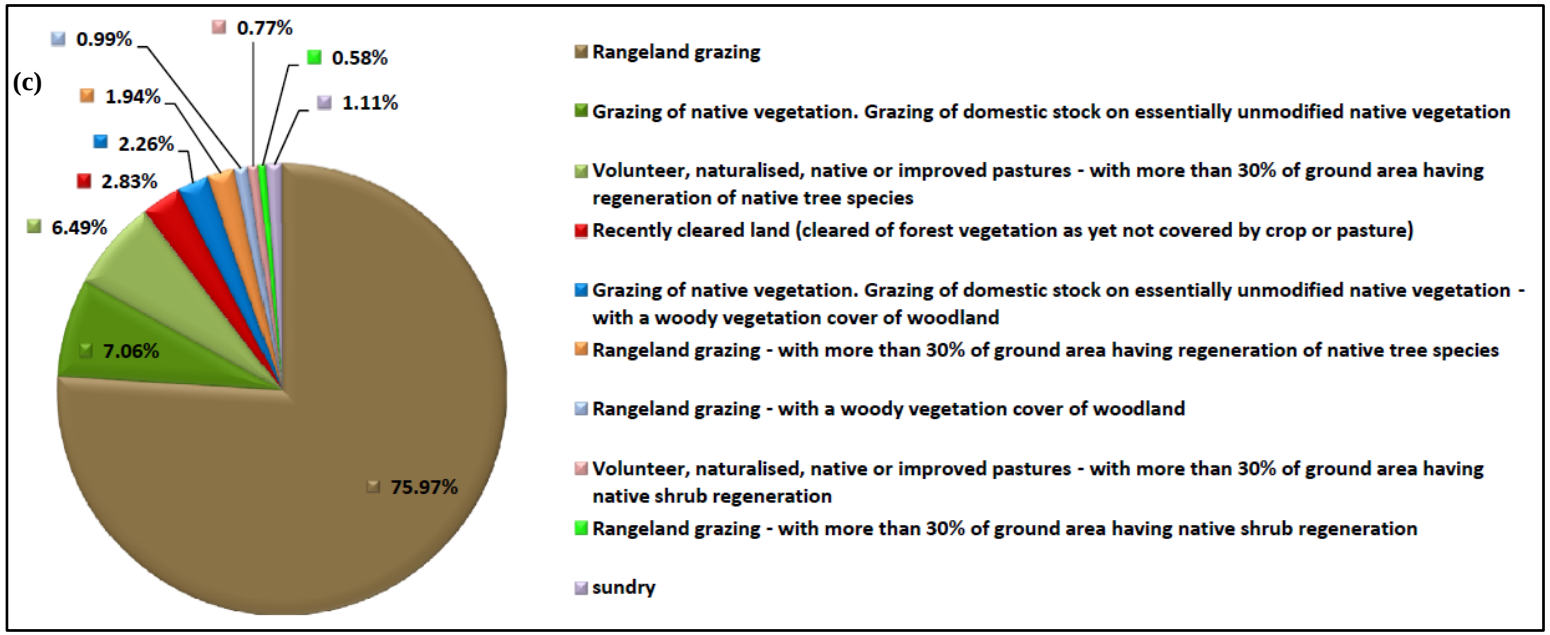


Figure A-V-15. Areas of NSW commercial rangeland deforested from 1989 to 2007 to: **(a)** all land uses, **(b)** infrastructure, and **(c)** pasture. Note that clearance to infrastructure, according to the LANDSAT classification does not match the LanduseV1 classification as one may expect, because the majority is actually rangeland grazing— this being mostly fence-line clearing for commercial grazing (and not for roads or railways etc.).

Table A-V-6. Sources of international data.

Region	Content	Source
world	biomes	World Wildlife Fund Terrestrial Ecosystems of the World Dataset (Olson et al., 2001)
world	rainfall, temperature	CRU TS 2.1 Climate Database, Consortium for Spatial Information: http://cru.csi.cgiar.org/ (New et al., 2002; Mitchell and Jones, 2005)
Canada	rangeland boundary	Land Cover of Canada 1995, version 1.1, Cihlar and Beaubien (1998)
Europe	rangeland boundary	Corine Land Cover dataset 2009, European Environment Agency , http://www.eea.europa.eu/data-and-maps/data/corine-land-cover-clc1990-250-m-version-12-2009
New Zealand	rangeland boundary	Land Resource Inventory, Landcare Research, Newsome et al. (2008)
USA	rangeland boundary	US Geological Survey land-use and land-cover dataset (2007) http://water.usgs.gov/GIS/metadata/usgswrd/XML/ds240_landuse_poly.xml
Russia	rangeland boundary	Land Resources of Russia dataset (2002), International Institute for Applied Systems Analysis and Russian Academy of Sciences. http://www.iiasa.ac.at/Research/FOR/russia_cd/guide.htm
Other	rangeland boundary	Russ Kruska & Robin Reid, International Livestock Research Institute (Nairobi), Reid et al. (2004), P.K.Thornton et al. (2002)

Table A-V-7. Biomes of NSW rangelands.

ID	Description	Area in Australia (Mha)	% of Australia	% in NSW	% of NSW Rangelands
8	Temperate shrublands, grasslands or savannahs	30.3	3.9	56	56
12	Mediterranean forests, woodlands, or scrub	15.0	1.9	46	22
13	Desert, or xeric shrublands	47.0	6.1	14	22
	Total	92.3	12	—	100

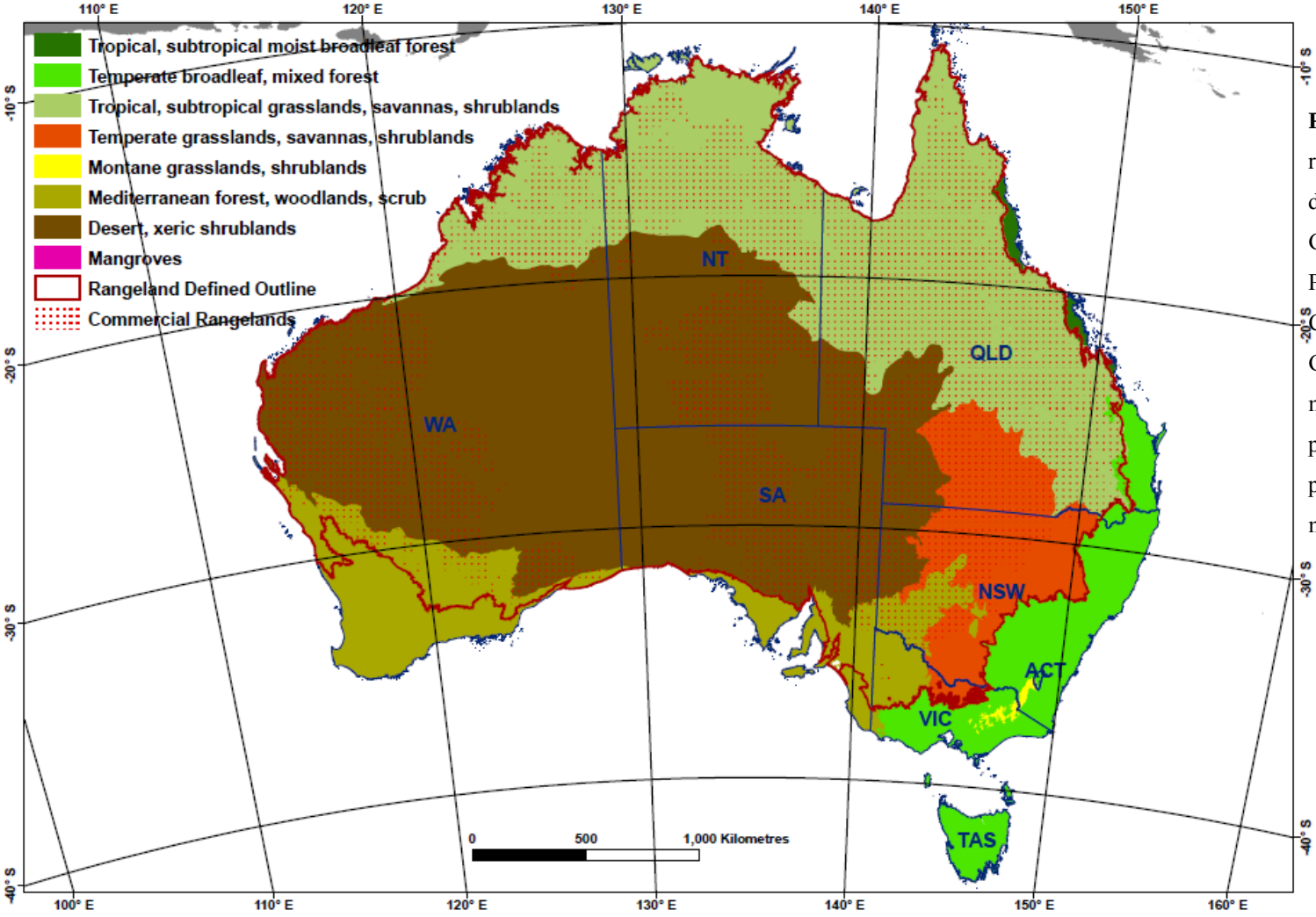


Figure A-V-16. Australian rangeland biomes, and outline of defined 'rangeland zone'. Biome GIS data from Olson et al. (2001). Projection = Lambert Conformal Conic, datum = Australian Geodetic Datum 1984, central meridian = 135° E, standard parallel 1= 18° S, standard parallel 2= 36° S, linear unit = metres.

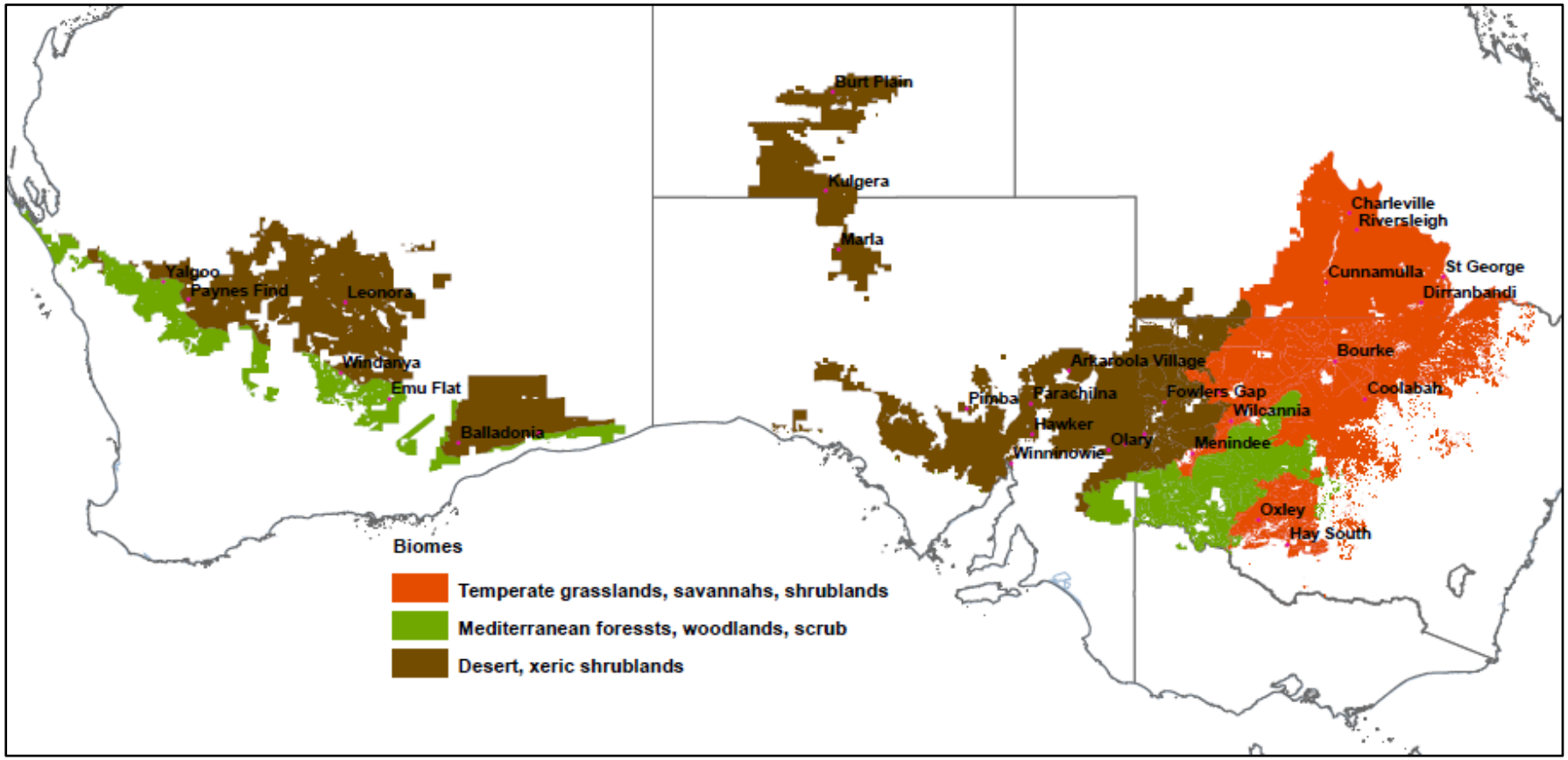


Figure A-V-16. Australian [commercial] rangelands with similar climate (rainfall and average temperature) to those of NSW. Geographic projection.

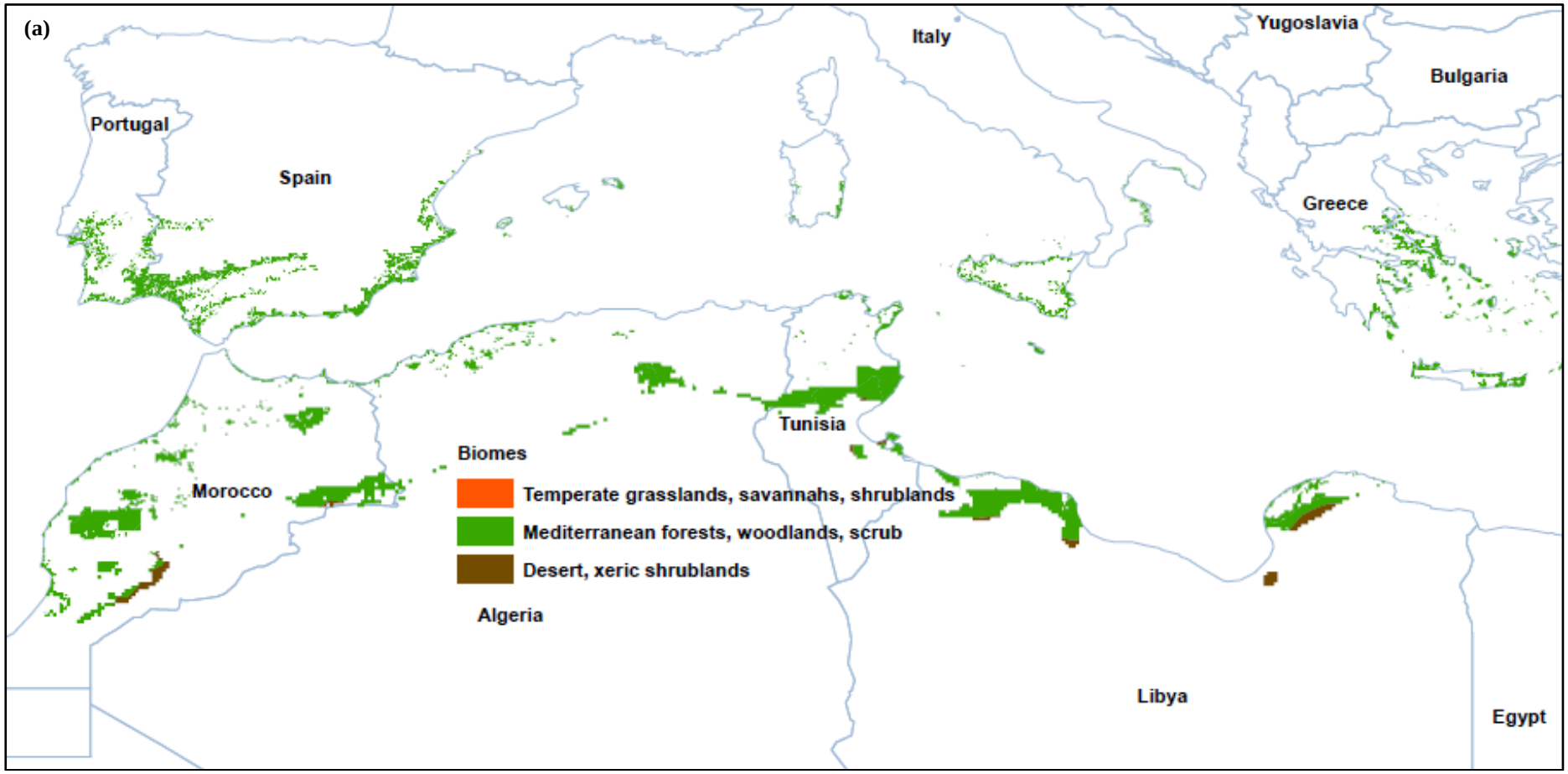
Table A-V-8. Global rangeland with biome 8 (Temperate shrublands, grasslands or savannahs) and with climate like NSW rangelands.

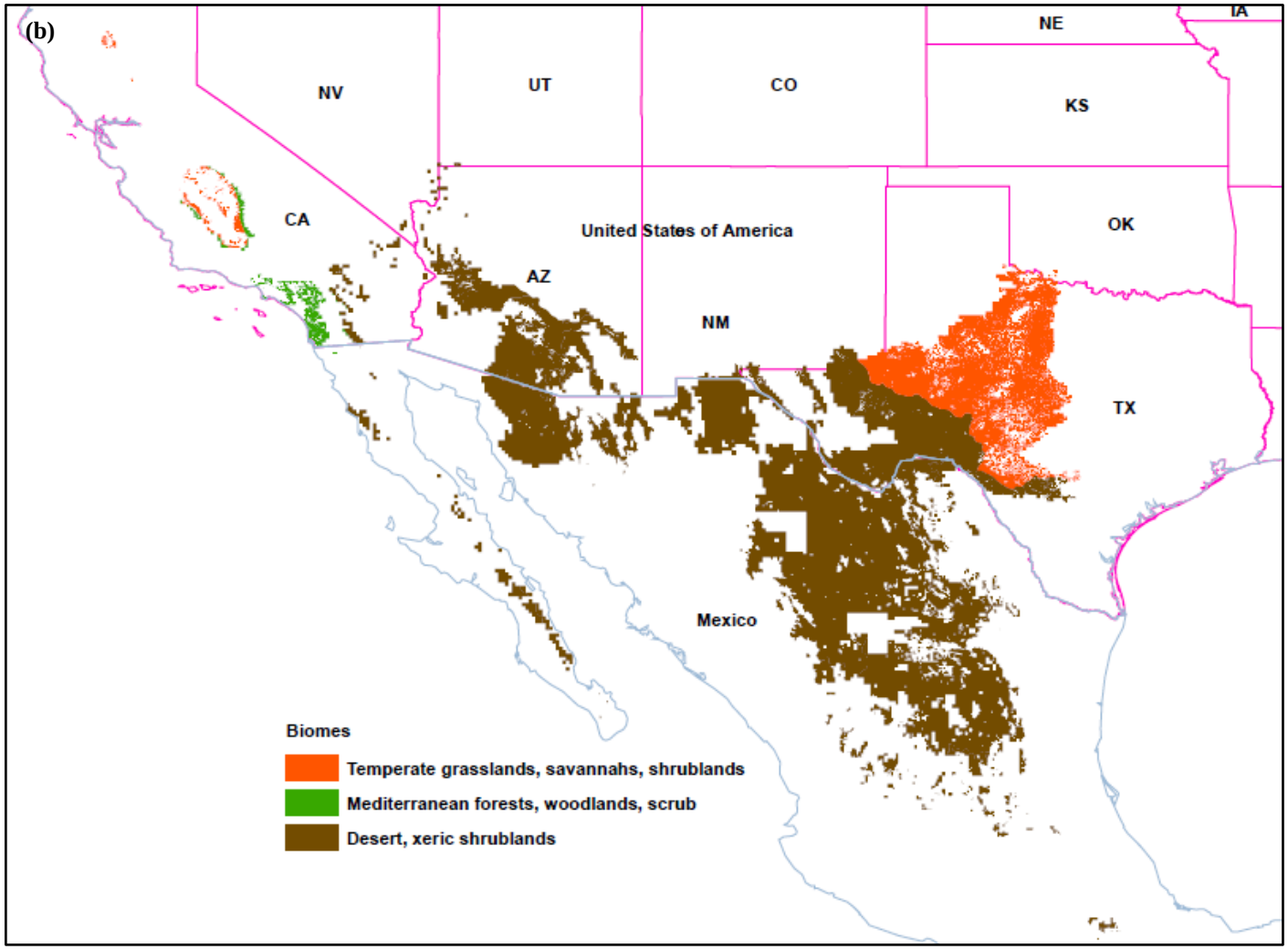
Country	Area (ha)	% of country/State
Australia	30,329,778	3.9
NSW	16,976,242	2.2
QLD	13,353,471	1.7
VIC	64	0.0000083
United States	7,792,779	0.85
Argentina	2,771,496	0.98
Syria	760,212	4.1
Iraq	509,967	1.2

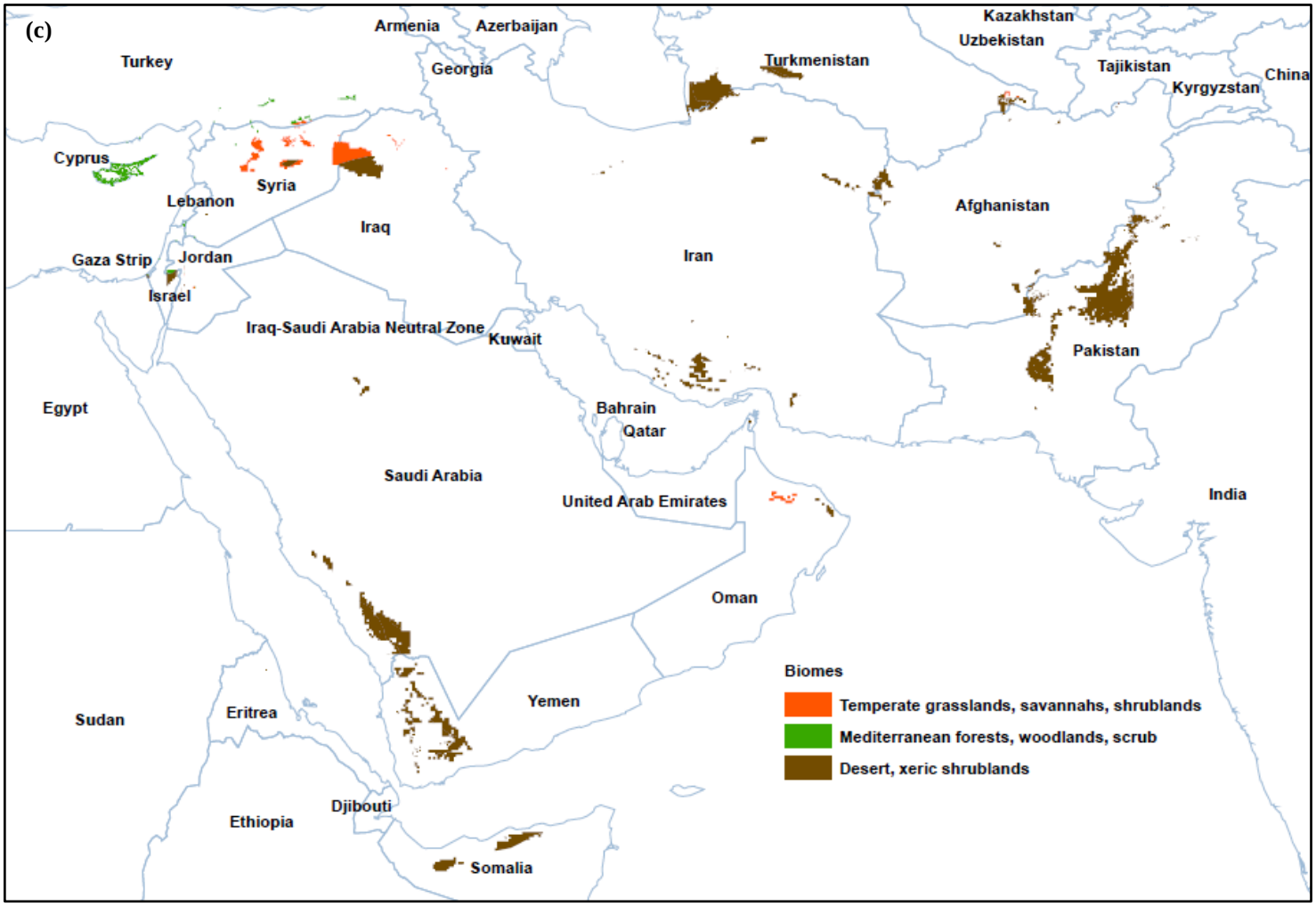
Table A-V-9. Global rangeland with biomes 12 and 13 and with climate like NSW rangelands. (12= Mediterranean forests, woodlands, or scrub, 13= Desert, or xeric shrublands)

Biome 12			Biome 13		
Country	Area (ha)	% of country	Country	Area (ha)	% of country
Ordered by area			Ordered by area		
Australia	14,953,771	1.9	Australia	47,017,892	6.1
WA	7130675	0.93	WA	18,571,286	2.4
NSW	6835555	0.89	SA	14,878,232	1.9
SA	942106	0.12	NSW	6,762,948	0.88
VIC	45434	0.0059	NT	5,812,547	0.76
Morocco	3,708,822	8.3	QLD	992,878	0.13
Libya	2.599.230	1.5	Mexico	31,956,843	16
Tunisia	1,985,139	13	Namibia	22,259,467	27
South Africa	1,508,599	1.2	South Africa	18,075,517	15
Algeria	1,252,027	0.53	Botswana	17,051,689	30
Spain	1,202,207	2.4	United States	12,295,248	1.3
United States	514,655	0.056	Pakistan	4,665,354	6.1
Greece	480,293	3.7	Turkmenistan	2,584,710	5.5
Cyprus	229,269	25	Saudi Arabia	2,113,365	0.98
Portugal	208,690	2.3	Iran	1,775,845	1.2
Italy	162,758	0.55	Yemen	1,739,319	3.3

Ordered by % of country			Ordered by % of country		
Malta	15,919	50	Botswana	17,051,689	30
Cyprus	229,269	25	Namibia	22,259,467	27
Tunisia	1,985,139	13	Mexico	31,956,843	16
Morocco	3,708,822	8.3	South Africa	18,075,517	15
Gaza Strip	2,140	5.9	Gaza Strip	4,289	12
Greece	480,293	3.7	Israel	149,507	6.9
Portugal	1,202,207	2.4	Australia	47,017,892	6.1
Spain	208,690	2.3	Pakistan	4,665,354	6.1
Australia	14,953,771	1.9	Turkmenistan	2,584,710	5.5
Libya	2,599,230	1.5	Yemen	1,739,319	3.3


(a)
Italy
Yugoslavia
Bulgaria
Portugal
Spain
Greece
Tunisia
Biomes
Temperate grasslands, savannahs, shrublands
Mediterranean forests, woodlands, scrub
Desert, xeric shrublands
Morocco
Algeria
Libya
Egypt


(b)
NV
UT
CO
NE
IA
KS
CA
United States of America
OK
AZ
NM
TX
Mexico
Biomes
Temperate grasslands, savannahs, shrublands
Mediterranean forests, woodlands, scrub
Desert, xeric shrublands


(c)
Armenia
Azerbaijan
Kazakhstan
Uzbekistan
Turkey
Georgia
Turkmenistan
Tajikistan
China
Kyrgyzstan
Cyprus
Syria
Lebanon
Iraq
Afghanistan
Iran
Gaza Strip
Jordan
Israel
Iraq-Saudi Arabia Neutral Zone
Kuwait
Pakistan
Egypt
Bahrain
Qatar
Saudi Arabia
United Arab Emirates
India
Oman
Yemen
Sudan
Eritrea
Djibouti
Ethiopia
Somalia
Biomes
Temperate grasslands, savannahs, shrublands
Mediterranean forests, woodlands, scrub
Desert, xeric shrublands

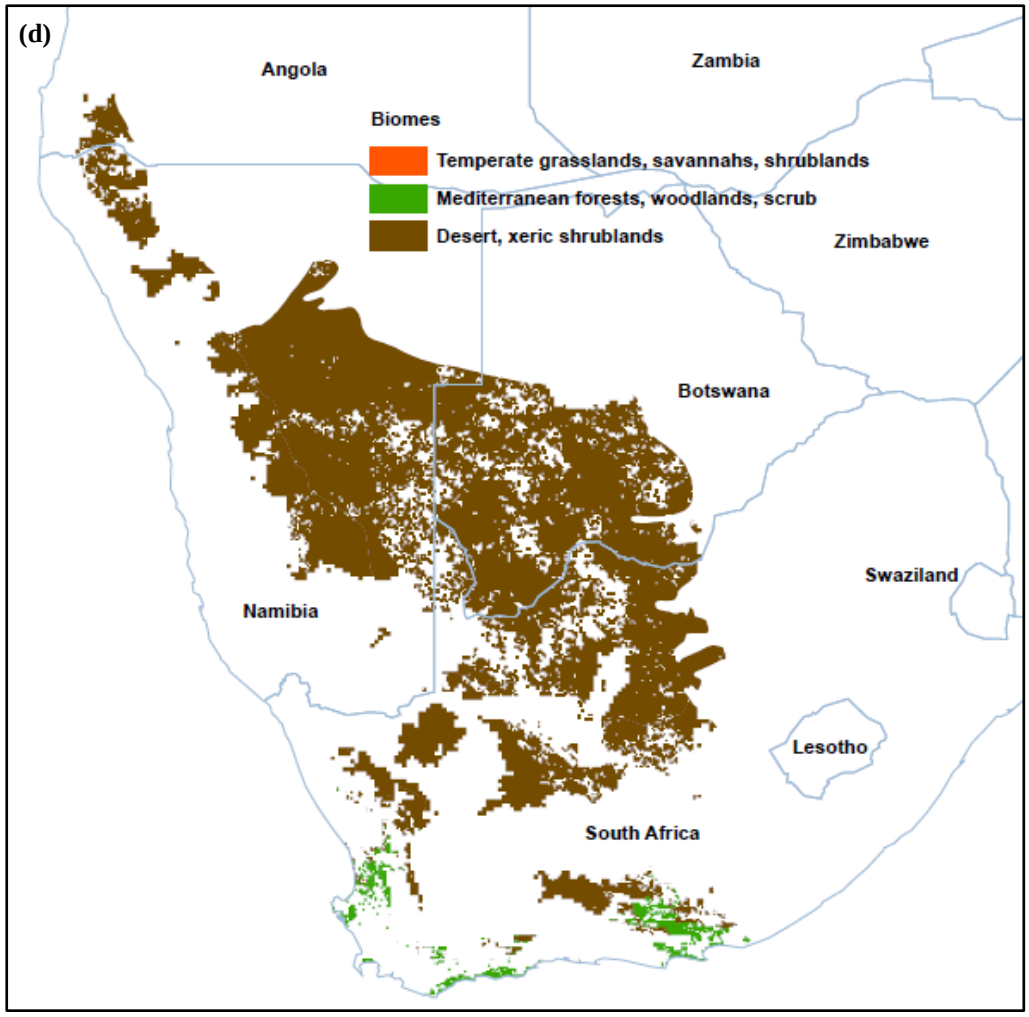


Figure A-V-17. Major global rangelands with the same biomes and similar climate (rainfall and average temperature) to those of NSW rangelands. Those in South America are more topologically distinct. **(a)** Mediterranean. **(b)** USA and Mexico **(c)** Middle East and neighbouring countries. **(d)** southern Africa.

Appendix VI. Taxonomy of tree species mentioned

Family	Genus	Species	Common name
Atherospermataceae	*Atherosperma*	*moschatum*	sassafras
Casuarinaceae	*Allocasuarina*	*decaisneana*	desert sheoak
Chenopodiaceae	*Atriplex*	*nummularia*	old man saltbush
Cupressaceae	*Sequoiadendron*	*giganteum*	giant sequoia
	Sequoia	*sempervirens*	coastal redwood
Malvaceae	*Brachychtion*	*rupestris*	bottle tree
Mimosaceae	*Acacia*	*aneura*	mulga
		cambagei	gidgee
		dealbata	silver wattle
		estrophiolata	southern ironwood
		harpophylla	brigalow
		melanoxylon	blackwood, blackwood wattle
Myrtaceae	*Corymbia*	*aparrerinja*	ghost gum
	Eucalyptus	*amygdalina*	black peppermint
		delegatensis, giganteum	gum-top stringy bark, alpine ash
		diversicolor	karri
		jacksonii	red tingle
		melanophloia	ironbark
		obliqua	stringy bark, messmate
		ovata	black gum, swamp gum, blue-leaved sally
		regnans	swamp gum, mountain ash
	Nothofagus, Lophozonia	*cunninghamii*	myrtle, myrtle beech
Phyllocladaceae	*Phyllocladus*	*aspleniifolius*	celery-top pine
Pinaceae	*Picea*	*rubens*	red spruce
		sitchensis	Sitka spruce
	Pseudotsuga	*menziesii*	Douglas fir
Rutaceae	*Nematoleposis*	*squamea* ssp. *squamea*	lancewood, satinwood